

EXHIBIT B



Expert Report

**Illinois River Watershed Water Quality and
Source Assessment**

Prepared for:

Illinois River Watershed Joint Defense Group

Prepared by:

Quantitative Environmental Analysis, LLC

Montvale, NJ

January 30, 2009

UNITED STATES DISTRICT COURT
FOR THE NORTHERN DISTRICT OF OKLAHOMA

STATE OF OKLAHOMA, ex. rel. W.A. DREW
EDMONDSON, in his capacity as ATTORNEY
GENERAL OF THE STATE OF OKLAHOMA
and OKLAHOMA SECRETARY OF THE
ENVIRONMENT, J.D. Strong, in his
capacity as the TRUSTEE FOR NATURAL
RESOURCE FOR THE STATE OF
OKLAHOMA,

Plaintiffs,

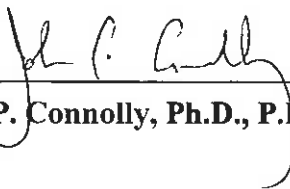
v.

TYSON FOODS, INC., TYSON
POULTRY, INC., TYSON CHICKEN, INC.,
COBB-VANTRESS, INC., AVIAGEN, INC.,
CAL-MAINE FOODS, INC., CAL-MAINE
FARMS, INC., CARGILL, INC., CARGILL
TURKEY PRODUCTION, LLC, GEORGE'S
INC., GEORGE'S FARMS, INC., PETERSON
FARMS, INC., SIMMONS FOODS INC., and
WILLOW BROOK FOODS, INC.,

Defendants.

Case No. 05-CV-329-GKF-SAJ

EXPERT REPORT OF


John P. Connolly, Ph.D., P.E., B.C.E.E.



January 30, 2009

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SECTION 2

POULTRY LITTER IS NOT A MAJOR SOURCE OF PHOSPHORUS TO THE ILLINOIS RIVER IN OKLAHOMA

2.1 SUMMARY OF DETAILED FINDINGS

- Naturally occurring phosphorus compounds, particularly dissolved inorganic phosphates available for algal growth, are the only forms of phosphorus at issue in this case.
- There are many contributors of phosphorus to the Illinois River and Lake Tenkiller.
- The pollutant fingerprints in the Illinois River and Lake Tenkiller do not match that of poultry litter.
- Poultry litter does not produce more phosphorus runoff than cattle manure or any other fertilizer applied intentionally or naturally (grazing cattle).
- Poultry litter does not produce more phosphorus than other applied fertilizer.
- Changes in water quality in Lake Tenkiller do not track with changes in poultry production.
- The similarity of water quality in Lake Tenkiller and other lakes in the region indicates that the use of poultry litter in the Illinois River Watershed does not degrade water quality beyond what occurs because of development for agriculture and urbanization and the nature of run-of-river reservoirs.
- Wastewater treatment plants appear to be the most important source of bioavailable phosphorus to the system.
- Lake sediment phosphorus is a minor source of bioavailable phosphorus.

2.2 NATURALLY OCCURRING PHOSPHORUS COMPOUNDS, PARTICULARLY DISSOLVED INORGANIC PHOSPHATES AVAILABLE FOR ALGAL GROWTH, ARE THE ONLY FORMS OF PHOSPHORUS AT ISSUE IN THIS CASE

Phosphorus is found in the environment as part of various inorganic and organic substances.¹ The inorganic substances include mineral phosphates and phosphate ions. The organic substances are biological residues. In the aquatic environment (i.e., lakes and rivers), some phosphorus-containing substances are dissolved in the water, others are particles. The particles include mineral phosphates, organic particles and inorganic particles to which phosphates have attached (Figure 2-1). Some forms of organic and inorganic phosphorus dissolve in the water. Dissolved organic phosphorus can be mineralized by bacteria to form dissolved inorganic phosphorus. This form of phosphorus, which is also called soluble reactive phosphorus, is consumed by algae for growth.

Dissolved inorganic phosphorus in lakes and rivers originates from several places: mineralization of dissolved organic phosphorus; dissolution of particulate inorganic phosphorus; or direct input from an outside source such as a wastewater treatment plant. Phosphorus is a micronutrient necessary for the growth of algae that form the base of the river and lake food webs. However, too much dissolved inorganic phosphorus under certain conditions may result in enough algae growth to cause aesthetic and dissolved oxygen problems.²

¹ Because phosphorus readily binds with other elements, phosphorus does not exist in its elemental form in the environment.

² Dr. Olsen (2008) in Section 6.4.3.5 of his report lists numerous “hazardous” substances allegedly found in poultry litter. Dr. Coale (2008) indicates that a number of the chemicals listed in Olsen (2008) are not commonly found in poultry litter (see Coale 2008; Opinion 31.) Also, the phosphorus referred to in Table 302.4 of 40 CFR § 302.4 is elemental phosphorus, which does not naturally occur in the environment. Consequently, of the chemicals remaining in Dr. Olsen’s list, only one, ammonia, can contribute to eutrophication. It’s generally accepted that phosphorus is the nutrient controlling eutrophication in the system; therefore, the focus of my report is on phosphorus. Ammonia and some of the other chemicals listed by Dr. Olsen can cause fish toxicity; however, none of the State’s assessments of the waters within the Illinois River Watershed indicate any issues with fish toxicity (see Section 6 for further discussion).

2.3 THERE ARE MANY CONTRIBUTORS OF PHOSPHORUS TO THE ILLINOIS RIVER AND LAKE TENKILLER

Drs. Cooke, Welch, Fisher, and Engel argue that phosphorus from poultry litter has degraded water quality in the Illinois River Watershed (Cooke and Welch 2008; Fisher 2008; Engel 2008). In doing so, they overstate the degree of water quality degradation and fail to properly account for the other sources of phosphorus in the watershed. Both point and non-point sources exist, as illustrated in the conceptual diagram in Figure 2-2. Livestock and application of commercial fertilizers are significant (see Sections 2.5 and 2.6), as are the sources that result from population growth, particularly the deforestation and urbanization associated with such growth (Grip 2008).

Deforestation is a well-documented cause for increased non-point source loadings to receiving waters. Forests tend to be relatively good “conservers” of sediment and nutrient loads. But, when forests are cut down for logging, development, or other purposes, the soils are less “protected” from erosion events and large sediment loads from deforested lands can occur. These sediment loads carry with them numerous constituents, including phosphorus. When cleared forest lands are developed, further increases in non-point source loading occur because development introduces impervious cover (e.g., parking lots, roads, rooftops, etc.). Schueler (2000) found that the first pulse of non-point source loadings due to urbanization occurred during the construction phases. Schueler (2000) also indicated that the influence of impervious cover carries throughout the lifetime of the urban environment. In fact, Schueler (2000) cites a second and possibly larger sediment pulse from streambank erosion, which is a result of increased storm peak flows and water volumes that occur when a watershed becomes urbanized.³ Urbanization also causes pollution from point source discharges of wastewater treatment plants (Brinkmann 1985).

Because urbanization degrades water quality, measures of urbanization are typically used to evaluate the potential impacts of development on water quality. Randhir (2003) indicated

³ Haraughty (1999) cites bank erosion along the Illinois River and its tributaries as a “substantial threat” to the Illinois River Watershed and indicates this erosion is likely due to degraded riparian areas, roads, and bridges. Grip (2009) also indicates that streambank erosion has occurred in the watershed.

“...impervious cover is widely used as an indicator of water resources degradation.” Horner et al. (1997) and Lindsey et al. (1997) concluded that impervious cover affects water quality and quantity and disrupts the natural ecosystem. Research has shown that even small percentages of urbanization impact water quality. The Randhir (2003) study found that sediment loadings were “especially high” from sub-basins with urbanization as low as 3%. Booth and Reinelt (1993), Horner et al. (1997), and May et al. (1997) all found that stream channels experience consistent bank erosion during storm events even when their contributing watersheds have relatively low impervious cover.

Besides non-point source pollution, the population increase in the Illinois River Watershed has increased pollution via septic systems (Sullivan 2009) and increased wastewater treatment plant discharges (Jarman 2008). Because much of the phosphorus load from a wastewater treatment plant is dissolved and bioavailable for growth of algae, the increased discharge from wastewater treatment plants can sometimes be more of a concern than the non-point source pollution caused by urbanization. Even if centralized wastewater treatment is not employed to deal with increasing populations, the increased number of septic systems can potentially contaminate groundwater (and subsequently, surface water).

Activities on the Illinois River Watershed other than poultry litter application impact water quality. Even small percentages of impervious cover (percentages that are seen in the Illinois River Watershed) can cause water quality degradation. Consequently, these activities cannot be ignored.

2.4 THE POLLUTANT “FINGERPRINTS” IN THE ILLINOIS RIVER AND LAKE TENKILLER DO NOT MATCH THAT OF POULTRY LITTER

According to the Plaintiffs’ consultants Drs. Fisher and Olsen, poultry litter is the primary source of the phosphorus in Lake Tenkiller and the streams within the Illinois River Watershed. They base this argument on their belief that poultry litter has a unique chemical signature that is maintained as runoff carries some of the chemicals in poultry litter from litter-applied land (LAL) through the watershed to the Illinois River and through the river to Lake

Tenkiller. However, they make no effort to demonstrate how chemicals in poultry litter would make their way across the land to local streams, down these streams to the Illinois River and through the Illinois River to Lake Tenkiller, nor what would happen to these chemicals along this journey. Such a journey is a complex adventure impacted by complex physics, chemistry, and biology. Along the way, chemicals are diluted, trapped, and transformed. Typically, complex fate and transport models are developed to account for all the processes at work. Drs. Fisher and Olsen ignored all of this. Dr. Engel made an attempt at modeling, but he too ignored many of the important processes and Dr. Bierman, in his January 2009 Expert Report on behalf of the Defendants, documents the failings of Dr. Engel's efforts.

Drs. Fisher and Olsen relied on the naive and untested assumption that the relative concentrations of the chemicals in poultry litter are immune to the complex physics, chemistry and biology; or, more simply, whatever happens to one chemical happens to all chemicals. The relative concentration pattern in poultry litter was treated as a chemical signature or fingerprint they compared to the chemical signature of water and sediment samples. Dr. Fisher relied primarily on four chemicals as the basis for this signature: phosphorus, zinc, copper, and arsenic. Using these chemicals, Dr. Fisher concludes that sediments of the Illinois River and Lake Tenkiller are a mixture of uncontaminated soil and poultry litter. Dr. Olsen used these and other chemicals, which he subjected to a Principal Components Analysis (PCA), in an effort to ascribe the source of the chemicals found at various points in the river. Relying principally on water column sample data, he concludes that poultry litter has impacted most locations sampled in the watershed.

I have examined the Illinois River Watershed water quality data in various ways to determine whether Drs. Fisher and Olsen are correct; i.e., there is a chemical signature in environmental samples that matches the chemical signature of poultry litter. I began by comparing the relative concentrations of the major cations and the major metals in various environmental media.

Figures 2-3 and 2-4a and b, respectively show the relative concentrations of the major cations calcium (Ca), sodium (Na), potassium (K) and magnesium (Mg; used by Dr. Olsen) and

the metals zinc (Zn), copper (Cu), and arsenic (As; used by Drs. Olsen and Fisher) for water purported to represent runoff from fields to which poultry litter had been applied and river and stream samples under base flow and high flow and Lake Tenkiller. Calcium and magnesium cause water hardness and are present in similar proportions in all types of samples, reflecting their abundance throughout the watershed at concentrations that yield water hardness in the range of 120 to 154 mg/L as calcium carbonate. (This hardness range is characteristic of streams in eastern Oklahoma; <http://water.usgs.gov/owq/hot.html>). Lake Tenkiller exhibits somewhat lower calcium concentrations, possibly due to precipitation of calcium carbonate during the summer. Overall, calcium and magnesium concentrations are driven by the mineral composition of the watershed soils and provide no means to track poultry litter or other sources. Looking at potassium and sodium, the edge-of-field (EOF) samples from alleged poultry litter amended fields are unique; they have on average about two times more potassium than sodium. The stream and lake samples show the opposite pattern; they have more sodium than potassium in a proportion that is similar among base flow, high flow, and lake samples. Thus, the cation data provide no evidence that poultry litter has impacted streams in the watershed or Lake Tenkiller.

Looking at the major metals, the dominant metal in poultry EOF samples is Cu, whereas it is Zn in all the river, stream, and lake samples. Copper accounts for 54% of the sum of these metals in EOF samples, but less than 14% in river, stream, and lake samples. This poor match exists in soil and sediment samples as well, as shown in Figure 2-5. Thus, the major metals data provide no evidence that streams in the watershed or Lake Tenkiller have been impacted by poultry litter. This conclusion is supported by more detailed examination of the data that I will now describe.

The mean ratio of total Zn to total Cu for samples of poultry litter, alleged poultry litter amended soils, EOF samples from alleged poultry litter amended fields, river water and sediments, and Lake Tenkiller water and sediments are shown in Figure 2-6. An analysis of variance (ANOVA; $F_{6,540} = 69.5$, $P < 0.0001$; Tukey HSD $\alpha < 0.05$) indicates statistical differences in the mean total Zn to total Cu ratio among some of these sample groups (compare letters in Figure 2-6, different letters indicate statistically different ratios). The mean total Zn to total Cu ratios in poultry litter and alleged poultry litter amended soil are 1.5 and 2.1

respectively, and these ratios are not statistically different from each other. The ratio in the poultry EOF samples is 3.5, statistically higher than poultry litter. The increasing concentration of Zn relative to Cu moving from the litter to the soil and to EOF water, and the statistical difference between litter and EOF samples suggests that there is an additional source of Zn in the soil and/or that the transport rate of Zn exceeds that of Cu. Illinois River Watershed river water and sediment, Lake Tenkiller water and sediment, and Illinois River Watershed soils not exposed to poultry litter (control soils) have total Zn to total Cu ratios between 5.0 to 8.9, all substantially higher than the poultry associated sample groups (Figure 2-6) and statistically different from the poultry groups.⁴

Further insight regarding the differences in metal composition between poultry litter and EOF can be gained by looking at individual samples. As shown in Figure 2-7, the water at the edge of litter-applied fields (triangular symbols) can have concentration ratios very different from litter (diamond symbols). The plots in this figure show the samples arranged in order of increasing concentration ratio and plotted according to the probability of having a lesser ratio. For example, half the values have a lesser ratio than the value plotted at 50%. The water running off the litter-applied field can have as little as 6 times as much phosphorus as Zn (i.e., a concentration ratio of 6) or as much as 200 times as much phosphorus as Zn. In contrast, excepting two outliers, the poultry litter samples all have ratios between 30 and 60. About 60% of the EOF samples have ratios in particulate matter below 30, suggesting that the runoff tend to have less phosphorus per unit Zn than the litter. The opposite is true for Cu; these eroded particles have up to 700 times more phosphorus than Cu, with about 60% of the samples having a phosphorus-to-copper ratio greater than the maximum measured in litter of 60.

There are also differences in the how Cu and Zn are transported to the EOF; much more of the Cu is dissolved (a median close to 90% versus a median of about 50% for Zn). This concurs with the fact that Cu tends to be bound to dissolved organic matter (DOM).

⁴ Total zinc and copper levels in Lake Tenkiller water were usually below detection limit (only 6 detects in 74 samples), therefore Lake Tenkiller water is excluded from the comparative statistical analysis.

The differences in chemical signature between poultry litter and EOF samples show that the efforts of Drs. Olsen and Fisher were doomed from the start. Their fundamental assumption of a largely invariant chemical signature is false. There is no unique chemical signature for poultry litter that can be traced through the watershed. Moreover, many potential sources of phosphorus contain the chemicals Drs. Olsen and Fisher used to track sources and most of these chemicals have unique behavior in the environment. Consider phosphorus, zinc, copper, and arsenic; the focus of Dr. Fisher and an important part of the focus of Dr. Engel. Each of these chemicals undergoes unique chemical reactions that govern the extent to which they can be taken up by plants, travel with flowing water, form insoluble compounds, and become attached to particulate matter. The concentration of metals in runoff depends on the soil properties (i.e., pH, clay content, presence of DOM) and application history (Arias et al. 2005). In general terms, arsenic is more mobile than Zn and Cu (Gupta and Charles 1999). Copper has a higher affinity for organic matter than Zn (Impellitteri et al. 2002). In particular, Cu is documented to preferentially bind to DOM (Han and Thompson 2003; Romkens et al. 2004; Arias et al. 2005; Lu and Allen 2001; Grassi et al. 2000; Karathanasis 1999). Therefore, Cu mobility is highly dependant on the abundance and composition of organic matter in the litter and soil and the extent to which organic matter is leached from the soil during runoff events. In a two-year study using lysimeters in Arkansas, Pirani et al. (2006) found that only 0.3% of the Cu added to the soil by poultry litter application leached, whereas 49% of the applied Zn leached. A similar result is suggested by the soil sampling performed by Gupta and Charles (1999). Arias et al. (2005) showed that absorption/desorption hysteresis is higher for Cu than for Zn, adding to the reduced mobility of Cu as compared to Zn. On the contrary, Keller et al. (2002) found completely different behavior for sewage sludge applied to soil. In this case, the concentration of Cu in runoff was higher than the concentration of Zn. The Cu was mainly found in dissolved form bound to DOM, which leached from the sludge. These studies show that the mobility of Cu and Zn are not the same and are also not directly proportional to the soil concentration in all cases.

In addition to the differences in the availability of the different compounds, their natural abundance differs in different soils. For example, Zn is present in control soils at concentrations similar to concentrations found in litter-amended soils (Gupta and Charles 1999;

Pirani et al. 2006). The average values for total Zn concentration in alleged LAL samples is 34 mg/kg, which is in the same order as the value reported by Gupta and Charles (1999) for control soils (20 to 34 mg/kg).

The site-specific data as well as the published studies demonstrate the futility of using ratios among phosphorus, zinc, and copper to trace the fate of phosphorus applied to fields as part of poultry litter. There is no unique ratio among phosphorus, zinc and copper that can be used to determine how much, if any, of the phosphorus in lake or stream sediments came from poultry litter. Unfortunately, Dr. Fisher, ignoring this fact, relied on gross data comparisons that obfuscate important differences and inconsistencies that exist when samples of litter, EOF water, and sediments are compared.

Dr. Fisher presents graphs that purport to show that the stream and lake sediments are a simple mixture of poultry litter and control soils (Figures 24 and 32 of Fisher [2008]). A close examination of these figures reveals the problems just discussed. The data exhibit tremendous variability and the sediment data fail to fall along the lines that supposedly reflect the mixing of poultry litter and control soils. In fact, the differences between the data and the mixing line, which are hard to see because of the scales of the graphs, are substantial. Relying on average concentrations, those differences are illustrated below.

Mixing control soil and poultry litter would generate the phosphorus, zinc, and copper concentrations shown as lines on the graphs in Figure 2-8. The lines start at the lower left at the composition of control soil and move up and to the right as the amount of poultry litter increases. The graphs also show the phosphorus, zinc, and copper concentrations measured in lake sediments. None of the lake sediment samples match the control soil-poultry litter mixture lines. In other words, the sediment is not a mixture of control soil and poultry litter. The source identification method used by Dr. Fisher provides no useful information about the contribution of poultry litter or any other pollutant source to lake or stream sediment.

2.5 POULTRY LITTER DOES NOT PRODUCE MORE PHOSPHORUS RUNOFF THAN CATTLE MANURE OR ANY OTHER FERTILIZER APPLIED INTENTIONALLY OR NATURALLY (GRAZING CATTLE)

According to Drs. Olsen and Fisher, runoff from poultry litter amended fields is more contaminated than runoff from fields with grazing cattle. To justify this assertion, Dr. Olsen used EOF samples and the results of a synthetic precipitation leachate procedure (SPLP). The conclusions derived from the analysis of both datasets are wrong as I explain below.

The EOF samples were collected during or shortly after a storm event at more than 80 sites that purportedly drained fields to which poultry litter had been applied. Dr. Olsen compared the mean values of these samples to the mean of two EOF samples taken from sites with grazing cattle. In addition to relying on a specious comparison of incompatible sample sizes (i.e., 80 to 2), he did not determine whether the difference in mean values was statistically significant (i.e., that it signified a real difference).

A thorough analysis shows that for many of the contaminants of interest, the EOF concentrations for cattle and poultry are not statistically different. Most importantly, there is no difference for phosphorus. Figure 2-9 presents a probability plot of the concentration of total phosphorus found in the EOF samples for poultry litter and grazing cattle fields. As can be seen in the plot, the phosphorus concentrations measured in poultry litter fields are comparable to concentrations measured in grazing cattle fields. It is impossible to conclude from these data that the poultry litter fields contribute more phosphorus to runoff than do grazing cattle fields.

Dr. Olsen uses the SPLP study to compare the “leaching potential” of poultry litter and cattle manure. However, this study is not relevant to what happens when it rains on fields containing poultry litter or cattle manure because it was conducted on samples poultry litter or cattle manure and not on amended soil samples. Once applied to the soil, poultry litter and cattle manure change in ways that modify the availability of phosphorus and trace metals. For that reason, the SPLP study on the litter and manure samples does not predict concentrations of phosphorus and metals in runoff. Dr. Olsen acknowledged this fact; in his report he states that the results from the SPLP study are maximum quantities and that, “as shown by Dr. Engel’s

calculations, the actual quantities leached in the environment are substantially less”. Dr. Olsen works around this fact by contending that the relative leaching potential is predictive of the relative concentrations likely to be found in runoff. This argument is wrong because the extent of leaching depends on how the environment affects the composition and characteristics of the litter and manure.

To validate his analysis, Dr. Olsen cites conclusions from a paper by Sauer et al. (1999) comparing runoff from plots amended with poultry litter and cattle manure and subjected to synthetic rain. Dr. Olsen highlights the conclusion that the poultry amended plots “provided at least six times the amount of each nutrient” than the cattle amended plots. However, the paper also states the following:

...since the amount of nutrients transported was proportional to the amount applied, losses from the dairy manure and urine treatment were influenced by the assumptions used in determining the amount of feces and urine to apply. Clearly, grazing intensity and waste deposition patterns create potential for large degree of spatial and temporal variability of nutrient runoff from grazed pastures. Further studies in this area are warranted, especially as the potential for nutrient runoff from applied poultry litter diminishes with time after application, whereas, grazing animals continue to deposit wastes on soil surface throughout the growing season.

In other words, the authors acknowledge that the study is not conclusive of the effect of poultry litter and cattle manure in a field situation due to differences in the applied amount of cattle manure and the effect of aging. Additionally, the study did not consider the effect of the cattle manure deposition patterns (i.e., in general in shaded areas and close to the water [Schomberg et al. 2000; Wells and Dougherty 1997]) and the reduction in soil permeability caused by the treading of the cattle (Wells and Dougherty 1997).

Most importantly, Olsen’s analysis of the Sauer study is incomplete because he did not include the fact that the study clearly shows the important effect that aging (soil-litter and soil-

manure interaction over time) can have in the runoff from different amendments. Sauer's study comprises two synthetic precipitation events, the first one day after the application of the wastes to the plots and the second two weeks later. During the first event, the nutrient runoff concentrations from fields amended with poultry litter was higher than from the cattle manure plot. However, during the second event the nutrient runoff concentrations from the poultry litter-amended and cattle-grazed fields were not significantly different, as shown by a statistical analysis in Sauer's paper. This result clearly demonstrates the effect that aging can have in only 14 days. The effect of aging on phosphorus runoff was specifically noted by Sharpley (1997) and Kleinman and Sharpley (2003). In particular, Sharpley studied 10 different poultry litter applied soils in Oklahoma. He found that the time elapsed between litter application and rainfall significantly affects the runoff concentration of nutrients and recommended avoiding litter application during periods of high rainfall probability. This recommendation is one of the current United States Department of Agriculture (USDA) best practices for poultry litter application (Sharpley 2006).

Dr. Olsen does not consider the wealth of published evidence that runoff water quality depends on many variables and no general conclusion can be drawn signaling one fertilizer as inherently worse than others. In fact, the concentration of chemicals in runoff water depends not only on the concentration of those chemicals in the applied manure, but also on manure properties, manure application methodology, soil properties of the field where the manure is applied, runoff hydrology of the area, management practices related to erosion control, animal access to water bodies, and the chemical element being considered. For example, Sharpley (2006) stated that up to 80% of the total phosphorus in runoff water can be controlled through best management practices. Based on available literature, the following paragraphs present an analysis of the relative impact of poultry litter and cattle manure to water quality in the Illinois River Watershed.

There have been extensive efforts to relate phosphorous concentration in soil and the dissolved reactive phosphorus (DRP) concentration in runoff water. Pote et al. (1999) found that runoff did not always correlate with soil test phosphorus (STP) but was well correlated with the water extractable phosphorus (WEP) content of the soils. Subsequently, Kleinman et al. (2002a)

and DeLaune et al. (2004) showed that the phosphorus concentration in runoff from a field amended with manure was correlated with the WEP of the applied product. The concentration of total phosphorus in poultry litter is typically about 4 times higher than in cattle manure. However, Bremer et al. (2008) and Kleinman et al. (2005) show that the WEP is in the same range. Kleinman et al. (2005) reported average values of WEP of 2.3, 3.2, and 4.0 g/kg for beef cattle, poultry (broilers), and dairy manures, respectively. Dr. Olsen, in Table 6-4-1 of his report, confirms this similarity, showing an average WEP value for 16 samples of poultry litter of 1.44 g/kg and values of 3.02 and 0.95 for 5 samples of fresh and dry cattle manure, respectively. Therefore, if all other conditions are the same, similar phosphorus runoff is expected from soils amended with either litter or manure.

Mass balances of WEP can be calculated for the Illinois River Watershed using the published results from Kleinman et al. (2005), Dr. Olsen's report (2008), and the manure production values from Dr. Clay's Expert Report (2008). Tables 2-1 and 2-2 summarize the mass balances and show that non-poultry livestock produce more WEP than poultry livestock. The mass balance constructed using Dr. Olsen's data (Table 2-1) shows that non-poultry sources contribute 68% more than poultry livestock. The difference is smaller using Kleinman's data (Table 2-2) but still shows that non-poultry sources contribute 14% more WEP than poultry sources. Both mass balances indicate a bigger input of WEP from non-poultry sources even though they significantly under-represent non-poultry sources because they do not include horses, sheep, and wildlife.

Table 2-1. Water extractable phosphorus mass balance in the Illinois River Watershed.

Livestock	Average Dry Matter (%)	Average WEP g/kg (dry basis)	Average WEP g/kg (as deposited)	Annual Manure Contribution Tons	Annual WEP Contribution Tons
Beef cow	25%	3.0	0.76	1,870,847	1,422
Dairy cow	15%	4.0	0.60	154,296	93
Swine	20%	9.2	1.84	362,331	667
				Total Non-poultry	2,182
Layers	58%	4.9	2.84	113,141	322
Broilers	82%	1.4	1.18	691,234	812
Turkey	75%	6.3	4.73	35,397	167
				Total Poultry	1,301

Notes:

Beef cow dry matter and WEP (dry basis) from Dr. Olsen report Table 6.4-1. It was assumed that 90% of the manure is dry and 10% is fresh (Olsen 2008, page 6-11).

Broilers dry matter and WEP (dry basis) from Olsen (2008).

All other values for Average Dry Matter and Average WEP (dry basis) were extracted from Kleinman et al. (2005).

Manure contribution data from Clay (2008).

Table 2-2. Mass of Water Extractable Phosphorus (WEP) generated in the Illinois Watershed using Kleinman et al. (2005) data.

Livestock	Average Dry Matter (%)	Average WEP g/kg (dry basis)	Average WEP g/kg (as deposited)	Annual Manure Contribution Tons	Annual WEP Contribution Tons
Beef cow	37%	2.3	0.85	1,870,847	1,592
Dairy cow	15%	4.0	0.60	154,296	93
Swine	20%	9.2	1.84	362,331	667
				Total Non-poultry	2,351
Layers	58%	4.9	2.84	113,141	322
Broilers	71%	3.2	2.27	691,234	1,570
Turkey	75%	6.3	4.73	35,397	167
				Total Poultry	2,059

Notes:

Average Dry Matter and Average WEP (dry basis) values were extracted from Kleinman et al., 2005.

Manure contribution data from Dr. Clay (2008).

2.6 POULTRY LITTER DOES NOT PRODUCE MORE PHOSPHORUS RUNOFF THAN OTHER APPLIED FERTILIZERS

As mentioned before, there have been many studies describing the phosphorus runoff from different kind of organic and inorganic fertilizers. The following paragraphs summarize

some of the relevant studies that show the runoff of phosphorus is not typically higher in poultry litter amended soils than in soils amended with other organic and inorganic fertilizers.

Hall et al. (1994) measured the runoff from fields amended with commercial inorganic fertilizers at the recommended soil test rate and poultry litter at 10 tons per ha (4 tons per acre), which is a typically recommended rate in published agronomic guidelines. They found that the total mass of nitrogen in the runoff was about 25% lower from the poultry litter amended fields than for the commercial fertilizer amended ones. The losses of phosphorus due to runoff were about the same for both fields. It is interesting to note that Dr. Engel calculated a maximum poultry litter application in the area of about 2.3 tons per ha (0.9 tons per acre), which is more than four times lower than the applied in Hall's study and well below the typical recommended values. These facts suggest that the application of poultry litter in the Illinois River Watershed has been below normal standards and that the expected nutrient runoff losses would be lower than using a commercial fertilizer to increase crop production. DeLaune et al. (2004) also obtained much higher phosphorus runoff values using commercial fertilizers (in this case triple superphosphate [TSP]). All the fertilizers in the DeLaune study were applied at the same rate (78 kg P/ha) and the soluble reactive phosphorus (SRP) runoff from TSP was more than 6 times higher than from poultry litter, likely due to the fact that most of the phosphorus in commercial fertilizers is WEP and therefore more available to runoff.

Similar results were obtained by Kleinman et al. (2002b) in a study of the application of fertilizer at a rate of 100 kg/ha of total phosphorus from 4 different sources (poultry manure, dairy manure, swine, and diammonium phosphate [DAP]) on three different soils. They found no statistical difference in runoff DRP concentration from soils subject to DAP, poultry, or swine manure application. Runoff from soils subject to dairy manure was lower for two of the soils and the same for one of the soils illustrating the aforementioned impact of soil properties on runoff of the applied product. It should be noted that the application refers to poultry manure and not poultry litter; poultry litter has a different behavior than manure (i.e., typically litter is dryer and has less WEP than manure [Vadas and Kleinman 2006; Sharpley et al. 2004]).

A 2004 study performed by Sharpley et al. focused on the impact of long-term application of poultry manure, poultry litter, swine slurry and dairy cattle manure on soil phosphorus concentration. This study included fields in Oklahoma, New York, and Pennsylvania that had been subject to continuous application of these materials as fertilizer for more than 10 years. The application of waste to these fields was based on total phosphorus and annually ranged between 75 and 150 kg/ha for poultry litter and dairy manure. Using the average total phosphorus concentration of poultry litter, this application rate corresponds to between 5 to 10 tons per ha of litter (2 to 4 tons per acre), which is within the recommended application range. The soil analyses performed by this study show that the soils amended with dairy manure have higher total phosphorus and WEP than the soils amended with poultry litter. Average WEP for dairy manure amended soils was 40.2 mg/kg, whereas for the poultry litter amended soils the average value was 20.8 mg/kg. As presented before, there is strong correlation between WEP contents of the soil and runoff so these values imply that the dairy manure amended soils analyzed in this study will have higher phosphorus runoff than the soils amended with poultry litter.

From the studies summarized in the previous paragraphs, it can be concluded that the potential runoff from a fertilized field will depend on many variables like soil characteristics, field management, rate, and history of application, etc. Therefore, it is erroneous to generalize that a certain fertilizer generates a bigger impact than others as they all have potentially the same impact if the application meets best management practices.

2.7 CHANGES IN WATER QUALITY IN LAKE TENKILLER DO NOT TRACK WITH CHANGES IN POULTRY PRODUCTION

Dr. Fisher attempted to use the phosphorus concentrations in dated sediment cores from Lake Tenkiller to infer the historical trend in phosphorus loading to the lake. He compared this trend to the trend in poultry house density and concluded that they match. Unfortunately, he made three mistakes that invalidate this comparison:

- he incorrectly dated the sediment cores, thus skewing the phosphorus concentration trends;
- he did not account for variations in sediment phosphorus caused by variations in the iron and aluminum content of the sediment rather than variations in phosphorus loading to the lake; and
- he ignored the fact that phosphorus levels in the lake sediments have not risen since the late 1980s and may be trending downward, indicating that the phosphorus loading to the lake has not risen despite increases in poultry population.

2.7.1 Dating of Sediment Cores

Six cores were collected from Lake Tenkiller in August 2005 and four were selected for geochronological and chemical analysis (a fifth was subjected to chemical analysis, only). The cores were sectioned into 2-cm intervals and analyzed for radionuclides, metals, and nutrients. Radionuclides were measured in an effort to determine the age of each section, i.e., the year or years each section was deposited. The goal was to generate a time history of the concentrations of phosphorous and various metals on sediments depositing in the lake. Soster (2005a, 2005b, 2005c, 2006) dated these cores using unsupported (excess) ^{210}Pb activities (measured indirectly by the analysis of its radioactive decay product ^{210}Pb) in the constant initial concentration (CIC) model. Excess ^{210}Pb was calculated by subtracting ^{214}Bi activity, a surrogate for supported ^{210}Pb . The CIC model assumes that all sediments in the core had the same activity of excess ^{210}Pb at the time of deposition (Cohen 2003). Dates of sediment deposition are calculated by fitting the decrease in ^{210}Pb activity with depth to a radioactive decay model. Cesium (C's)-137 was used as an independent means of dating the sediment. The peak ^{137}Cs activity is associated with sediments deposited around 1963 and the first appearance of ^{137}Cs is presumed to represent about 1954.

The ages and sedimentation rates estimated by Soster (2005a, 2005b, 2005c, 2006) and presented by Dr. Fisher, which were calculated from the ^{210}Pb data, put the peak ^{137}Cs activity in sediment deposited much later than 1963. The differences are substantial, as shown in Table 2-3.

Table 2-3. The ^{210}Pb age as calculated by Dr. Fisher for the ^{137}Cs peak (presumed to be 1963).

Core	Year of Deposition of Sediments with the Peak ^{137}Cs Activity, as Assigned by Dr. Fisher	Rounded Difference Between Dr. Fisher's Date and 1963, the Expected Year When Depositing Sediments Would Have Maximum ^{137}Cs
LKSED-01	1970	7 years
LKSED-02	1971	8 years
LKSED-03	1977	14 years
LKSED-04	1982	19 years

Dr. Fisher argues that “concordant ages cannot be obtained for ^{210}Pb and ^{137}Cs methods” due to their differing input signals. This is not correct. The dating derived from ^{210}Pb and ^{137}Cs should be roughly consistent. The two dating methods are almost always used in combination. The ^{137}Cs profiles look good. They have defined ^{137}Cs peaks (~1963) near the bottom of the cores, which makes sense based on the fact that sedimentation would have begun in 1954 after the dam was completed. There is no reason to discount the ^{137}Cs data and therefore the significant differences in the dates determined using the two methods raise doubts about the ^{210}Pb dating analysis.

As seen in Figures 2-10a and 2-10b, two things stand out in the excess ^{210}Pb activity profiles for the Lake Tenkiller cores: 1) the ^{210}Pb activity in the top 6 to 10 cm of the cores is variable, but does not consistently decline with depth, likely due to biological mixing; and 2) the ^{210}Pb activity is relatively constant over the bottom 6 to 10 cm of the cores profile, suggesting rapid deposition of sediments when the lake was first created (i.e., these bottom sediments were all deposited at about the same time). Consequently, the top and bottom portions of the core do not conform to the assumptions of the CIC dating model and should not be used in determining the rate of sedimentation that is the basis for dating the core sections. The analysis was redone excluding the surface and bottom samples indicated by red circles around the data points in Figures 2-10a and b. The log-linear regression of the data from which the dating was determined (i.e., with the surface and bottom sample data points removed) is shown in Figure 2-11. The ages calculated by using only the excess ^{210}Pb activities for the mid-portions of each core coincide closely with the ^{137}Cs age for cores LKSED-02, -03, and -04 (Table 2-4). A discrepancy remains for core LKSED-01. For this core the 38-42 cm sample was pre-dam

closure Illinois River Floodplain sediment and provides a reference point of 1954, roughly when the lake began to fill in. The exact depth of the ^{137}Cs peak in this core is uncertain and falls between 30-36 cm. If we assign 1954 to the 40 cm depth in the core, the range of dates for the 30-36 cm section are 1960-1965, coinciding with the ^{137}Cs peak of 1963. Dating this core using ^{210}Pb activity results in much younger ages for the bottom sediment, therefore I used the ^{137}Cs dating results.

Table 2-4. The ^{210}Pb age as calculated by QEA for the ^{137}Cs peak (presumed to be 1963).

Core	Year of Deposition of Sediments with the Peak ^{137}Cs Activity, as Assigned by QEA	Rounded Difference Between QEA Date and 1963, the Expected Year When Depositing Sediments Would Have Maximum ^{137}Cs
LKSED-01	1965	0 years*
LKSED-02	1964	1 year
LKSED-03	1964	1 year
LKSED-04	1974	11 years

**This core was dated based on ^{137}Cs*

2.7.2 Proper Normalization of Sediment Phosphorus Concentrations

Sediments are composed of particulate organic matter, mineral particles, and precipitates. Phosphorus can be present as part of the particles themselves or sorbed to the particles. Particles with a greater amount of iron, aluminum, and calcium tend to have a greater amount of phosphorus. This is because some of the compounds that make up the sediment are combinations of phosphorus and these elements and because phosphorus readily adsorbs to iron and aluminum oxides and hydroxides that are common components of the sediment (Shukla et al. 1971; Detenbeck and Brezonik 1991; Novak and Watts 2006).⁵ As a result, the phosphorus concentration of a sediment sample depends on the nature of the sediment. This means that differences in phosphorus concentration alone cannot be used to infer differences in phosphorus loading.

The influence of iron and aluminum on phosphorus concentration is illustrated by examining a few representative stream sediment and control pasture soil samples collected by the

⁵ Phosphorus concentration is determined to a lesser extent by other constituents of the sediment, typically the most important of which is organic matter.

Plaintiff's consultants in 2005 and 2006. Table 2-5 and Figure 2-12 show for these few samples the concentrations of total phosphorus, Iron (Fe), aluminum (Al), and the ratio of total phosphorus to the sum of Fe and Al. The total phosphorus concentrations range from 123 to 775 mg/kg, about a factor of six; close to the range of total phosphorus concentrations found in stream sediments (111 to 921 mg/kg by method SW6020B). Note that the samples also have a wide range of Fe + Al concentrations that vary by almost a factor of five and that the total phosphorus concentrations correlate with the Fe + Al concentrations. To account for this correlation, the total phosphorus concentrations were normalized by the sum of Fe and Al (Daskalakis and O'Connor 1995). The normalized concentrations are all very similar, ranging over a factor of two, despite the fact that the total phosphorus concentrations range over a factor of six. Most significantly, the normalized concentrations for the stream samples fall within the range of values for the three control soil samples. In fact, of the many stream samples taken by the Plaintiffs' consultants, only one has a normalized total phosphorus concentration substantively above what was found for the control pasture soils; Station SD-008 (which is not shown in Table 2-5 or Figure 2-12) had a value of 0.044. The striking conclusion from this illustration is that higher concentrations of total phosphorus are not presumptive evidence of an anthropogenic source. These higher concentrations may simply reflect the greater presence of iron and aluminum compounds (or calcium compounds) that naturally contain phosphorus or have the ability to bind phosphorus naturally present in the environment. In other words, there is no evidence that the total phosphorus concentrations in stream sediments are higher than expected from soils running off of control fields. The total phosphorus concentrations differences among the stream sediment samples are the result of differences in the concentrations of Fe and Al in the material settling to the bottom.

Table 2-5. Phosphorus, iron, and aluminum content of a few representative stream and control soil samples.

Sample	Type	Location	Fe (mg/kg)	Al (mg/kg)	Fe + Al (mg/kg)	TP (mg/kg)	TP/ (Fe + Al)	Measurement Basis
SD-029	Sediment	Evansville Creek	4,668	1,383	6,051	123	0.02	Wet Weight
SD-203 ¹	Sediment	Evansville Creek	7,904	2,916	10,820	275	0.025	Wet Weight
CL2-B-4	Soil	Nickel Preserve	11,500	6,850	18,350	409	0.022	Unknown
CL2-B-2	Soil	Nickel Preserve	10,800	6,470	17,270	475	0.028	Unknown
CL2-A-2	Soil	Nickel Preserve	12,800	6,690	19,490	518	0.027	Unknown
SD-001 ¹	Sediment	Buddy Kidd Creek	19,326	7,869	27,195	775	0.028	Wet Weight

¹Average of 2 replicate samples.

An overall examination of total phosphorus concentration in relation to Fe + Al concentration shows a strong correlation for stream and lake sediments (Figure 2-13). Moreover, these two types of sediment exhibit remarkably similar correlations as evidenced by their plotting on top of each other. Control soil from a field matches this relationship. Poultry litter, which is also shown in Figure 2-13, does not. Strong correlations between phosphorus extracted from sediments and the coextracted iron plus aluminum have been demonstrated in a number of studies (e.g., Danen-Louwerse et al. 1993; Zhou et al. 2005).

The strong association of phosphorus with iron and aluminum compounds is documented for Lake Tenkiller sediment by measurements of the phosphorus associated with readily extractable Fe, Al, and Ca. These analyses were conducted by the Plaintiffs' consultants on 3 segments of 4 sediment cores collected from the riverine (LKSED-4), transitional (LKSED-3) and lacustrine (LKSED-2 and LKSED-1) regions of the lake. The data from these analyses, which are shown in Table 2-6, illustrate the importance of Al and Fe compounds as accumulators of phosphorus and their dominance over Ca compounds (together Al and Fe compounds account for six times more phosphorus than Ca compounds).

Table 2-6. Lake Tenkiller sediment phosphorus that is loosely-bound or associated with extractable Al, Ca, or Fe.

Sample	Al Bound P (mg/kg dry)	Ca Bound P (mg/kg dry)	Fe Bound P (mg/kg dry)	Loosely Bound P (mg/kg dry)
LKSED-4-01-01	124	46.3	232	2
LKSED-4-02-01	199	56.1	140	2
LKSED-4-03-01	246	65.6	121	2.61
LKSED-3-02-01	168	46.5	139	2
LKSED-3-03-01	186	55.2	118	2
LKSED-3-04-01	168	54	166	2
LKSED-2-02-01	164	46.8	126	4.51
LKSED-2-03-01	165	53.3	118	2
LKSED-2-04-01	155	52.3	149	2
LKSED-1-02-01	143	65	230	2
LKSED-1-03-01	168	71.2	310	2
LKSED-1-04-01	174	66.3	314	2.13
Average	172	57	180	2

Total phosphorus concentrations in the four lake sediment cores vary with depth as shown in the top row of plots shown in Figure 2-14. Much of this variability disappears when the concentration is normalized by the Fe + Al concentration, as shown in the bottom row of plots in the figure. The normalized concentrations provide a good sense of the trends in total phosphorus loading to the lake over the period over which the sediments accumulated on the lake bottom.

2.7.3 Time Trends in Lake Tenkiller Phosphorus

Using the corrected dating of the segments in the lake sediment cores and the Fe + Al normalized total phosphorus concentrations, the time trend of Lake Tenkiller phosphorus is shown in Figures 2-15 and 2-16. Sediments deposited prior to the mid-1960s had normalized total phosphorus levels of about 0.020 to 0.025 g total phosphorus per g Fe + Al, which are within the range of levels found in field control soils and most stream sediments (i.e., 0.020 to 0.028 g total phosphorus per g Fe + Al). The most upstream core (LKSED-04) shows an

increase from the late-60s to the early-80s to about 0.035 g total phosphorus per g Fe + Al followed by a slow decline to about 0.031 g total phosphorus per g Fe + Al in 2005. This upstream core gives the best indication of the trends in total phosphorus load to the lake. The next downstream core (LKSED-03) exhibits less change over time, with concentrations throughout the core remaining in the range seen for field control soils (0.020 to 0.028). The further downstream cores are impacted by phosphorus cycling that occurs due to lake thermal stratification and depletion of oxygen in the hypolimnion. As a result, these cores show a somewhat different time pattern that shows a more gradual increase in concentration to a peak in the late-1980s at about 0.030 to 0.035 g total phosphorus per g Fe + Al and relatively constant concentrations to 2005. These temporal trends concur with Litke (1999) who reports that phosphorous concentrations are decreasing in many National Water-Quality Assessment Program (NAWQA) study units.

The fact that total phosphorus load to the lake reached a maximum in the 1980s and has remained relatively constant or declined slightly since that time is inconsistent with the hypothesis that poultry litter is a dominant source of the total phosphorus in the lake. As shown by Dr. Fisher, the poultry population in the watershed increased substantially over the period between the 1980s and 2005. By his estimate, the population was about 465,000 animal units in 1982, 688,000 in 1992, and 850,000 in 2002 (Smith 2008). Thus, it almost doubled over this 20-year period and increased by about 24% in the last 10 years. Assuming the poultry litter total phosphorus contribution to the lake has been proportional to the poultry population, lake sediment total phosphorus would have risen significantly over the last 20 years if poultry litter was an important total phosphorus source. The absence of a rise means that poultry litter cannot have been an important source.

Another interesting result of the above analysis is that it indicates that in the absence of anthropogenic phosphorus sources the sediments of the river and lake would have about 0.020 to 0.028 g total phosphorus per g Fe + Al. Thus, anthropogenic sources are responsible at most for about 0.01 g total phosphorus per g Fe + Al (i.e., 0.035 minus 0.025), which constitutes about one-third of the phosphorus in the sediments.

2.8 THE SIMILARITY OF WATER QUALITY IN LAKE TENKILLER AND OTHER LAKES IN THE REGION INDICATES THAT THE USE OF POULTRY LITTER IN THE ILLINOIS RIVER WATERSHED DOES NOT DEGRADE WATER QUALITY BEYOND WHAT OCCURS BECAUSE OF DEVELOPMENT FOR AGRICULTURE AND URBANIZATION AND THE NATURE OF RUN-OF-RIVER RESERVOIRS.

The Plaintiffs' consultants contend that poultry litter is the principal cause of water quality problems in the Illinois River Watershed. While they acknowledge the presence of other sources of nutrients and bacteria, they conclude that water quality problems would be minimal in the absence of poultry litter application as a fertilizer. If this conclusion is correct, one expects other reservoirs in nearby watersheds that have much less poultry litter application but similar land use to have better water quality. To test this hypothesis, the water quality of other lakes in the state that have watershed characteristics similar to the Lake Tenkiller watershed, but lower poultry populations, were compared to the water quality of Lake Tenkiller.⁶

Lakes Hugo and Sardis, which are in southeastern Oklahoma (Figure 2-17), were found to be the best available comparisons to Lake Tenkiller. All are in the same general physiographic region of the southern Midwest and contain portions of the Arbuckle and Ozark mountain chains, which are characterized, at least in part, by the presence of karst features including caves and conduits to groundwater recharge, flow, and discharge. The land use distributions of the three watersheds are summarized in Table 2-7. The Tenkiller watershed is the most developed and contains the most pasture and hay. All three have a large fraction forested. No records indicated extensive silviculture or industrial activities in any of the watersheds.

⁶ Although there is no one "perfect" comparison to Lake Tenkiller that has *all* of the same characteristics, but little or no poultry within the basin, attempts were made to find comparable watersheds that had important characteristics similar to that of the Lake Tenkiller watershed.

Table 2-7. Land use distribution for Lakes Tenkiller, Hugo, and Sardis Watersheds.

Land Cover	Tenkiller	Hugo	Sardis
Open Water	1.5%	2.6%	8.0%
Developed Open Space	5.6%	2.9%	1.5%
Developed, Low Intensity	2.1%	0.3%	0.1%
Developed, Medium Intensity	0.7%	0.1%	0.0%
Developed, High Intensity	0.3%	0.0%	0.0%
Barren Land	0.1%	0.1%	0.0%
Deciduous Forest	41.5%	33.7%	44.8%
Evergreen Forest	1.2%	23.3%	12.8%
Mixed Forest	0.5%	7.4%	9.0%
Shrub/Scrub	0.5%	1.6%	1.6%
Grassland/Herbaceous	3.4%	8.6%	5.6%
Pasture/Hay	42.0%	18.0%	15.4%
Cultivated Crops	0.1%	0.2%	0.0%
Woody Wetlands	0.6%	1.2%	1.1%
Emergent Herbaceous Wetlands	0.0%	0.1%	0.0%

Land cover information from 2001 Multi-Resolution Land Cover dataset.

An important characteristic pertinent to the comparison is the ratio of the size of the watershed to the size of the reservoir. This watershed to water surface area ratio is a measure of the area of land contributing runoff per unit area of reservoir. A higher value means more potential for water quality issues because relatively more land is contributing nutrients and bacteria to the lake. Given the importance of the watershed to water surface area ratio, comparisons to Lake Tenkiller need to be made in light of differences in these ratios.⁷ The watershed to lake area ratios of Lakes Tenkiller, Hugo, and Sardis are 82.3, 81.4, and 12.3, respectively (Table 2-8). These ratios indicate that Tenkiller and Hugo undergo similar areal loading, while Sardis experiences significantly less.

⁷ It should be noted that Drs. Cooke and Welch identify the watershed to water surface area ratio as an important differentiation between lakes and run-of-the-river reservoirs on page 9 (last paragraph) of their report. However, they ignore this characteristic when comparing Broken Bow to Lake Tenkiller. Broken Bow has a watershed to water surface area ratio of 37, while Lake Tenkiller's is 82. See Horne (2009) and Sullivan (2009) for further discussion concerning the inappropriateness of using Broken Bow as a comparative watershed to Lake Tenkiller.

Table 2-8. Comparison of various watershed characteristics among Lakes Tenkiller, Hugo, and Sardis.

Lake	Storage Conservation Control Pool (acre-ft)	Watershed Area (acre)	Water Surface Area (acre)	Watershed/ Water Surface Area Ratio	Average Depth (ft)
Lake Tenkiller	654,100	1,052,800	12,800	82.3	51.1
Lake Hugo	166,954	1,093,760	13,440	81.4	12.4
Lake Sardis	274,333	167,040	13,610	12.3	20.2

Hugo and Sardis have fewer poultry counts per unit area than Tenkiller (Table 2-9). The Tenkiller watershed contains approximately 213 animal units of poultry per square mile. The Hugo and Sardis watersheds contain seven and less than one animal units per square mile, respectively. The cattle populations in the Tenkiller, Hugo, and Sardis watersheds are 42, 28, and 25 animal units per square mile and the swine populations are 7, 2, and 6 animal units per square mile, respectively. The Tenkiller watershed contains the greatest density of people; 163 per square mile compared to 12 and 8 in Hugo and Sardis, respectively.

Table 2-9. Population counts for the Tenkiller, Hugo, and Sardis watersheds.

Lake	Active Poultry Houses per sq mi	2002 Cattle Animal Units per sq mi ¹	2002 Swine Animal Units per sq mi ¹	2005 Human Population per sq mi
Lake Tenkiller	1.2 (1.1) ²	106	18	163
Lake Hugo	0.02	28	2	12
Lake Sardis	<0.01	25	6	8

Notes:

¹. Poultry, cattle, and swine animal units acquired through personal communication with Raleigh Jobes.

². Number of active poultry houses per Plaintiffs' consultant J. Berton Fisher. Number of active poultry houses per defendants' contract growers in parentheses.

A review of USEPA Water Discharge Permits revealed no significant point-source dischargers in either the Hugo or Sardis watersheds. Point-source dischargers are direct contributors to the nutrient loads in a watershed. The absence of significant discharges not only eliminates anomalous nutrient sources in the comparative watersheds, but further supports the assertion that there are comparable or less human populations and industry in the Hugo and Sardis watersheds as compared to the Illinois River Watershed.

Run-of-the-river reservoirs typically have lacustrine, transitional, and riverine zones. Generally, these different zones have differing water quality. Hugo and Sardis are somewhat unique in that the transition from river to lake occurs over a short distance and these lakes lack the type of riverine zone seen in Tenkiller (Figures 2-18a through 2-18c). When comparing the water quality among these three lakes, it is critically important that comparisons are made for like sections.

A reservoir's residence time can influence water quality. If the residence time is short (i.e., less than about two months; Baker and Dycus 2006), the loss of phytoplankton with the water flowing out of the reservoir can limit the maximum phytoplankton concentration in the reservoir. Table 2-10 shows an estimate of the residence times of the three reservoirs. Because Lake Hugo's residence time is low enough to potentially impact maximum phytoplankton concentrations (i.e., maximum chlorophyll-a concentrations), it needs to be considered when comparing Lake Hugo to the other two reservoirs.

Table 2-10. Estimate of residence times for Lakes Tenkiller, Hugo, and Sardis.

Reservoir	Age (yrs)	Volume ^a (ac-ft)	Average Depth (ft)	Approx. Long-Term Average Inflow (cfs)	Period of Record	Estimated Residence Time (months) ^b			
						Whole Lake	Riverine	Transitional	Lacustrine
Hugo	33	157,700	11.9	2,100	1995-2007	1.3	0.1	0.4	0.7
Sardis	27	274,330	20.2	325	1995-2007	14.2	-	2.5	11.7
Tenkiller	56	654,100	50.7	1,245	1997-2007	8.8	0.3	1.3	7.3

^a At normal pool elevation.

^b At long-term average inflows.

Long term average inflow for Hugo, Sardis, and Broken Bow determined from United States Army Corps of Engineers (USACE) charts, for Tenkiller, used average United State Geological Survey (USGS) flows for Baron Fork, Caney Creek, and Illinois River at Talequah.

Water quality was compared in two ways. The phytoplankton concentrations, measured as chlorophyll-a, total phosphorus, and SRP concentrations in the upstream sections provide some sense to the potential impact of point and non-point sources of phosphorus in the watershed. The chlorophyll-a, total phosphorus, and SRP concentrations, dissolved oxygen

profiles in the lacustrine sections, and chlorophyll-a Trophic State Index (TSI) values provide evidence of the water quality impacts resulting from the watershed loads.

Figure 2-19 and Table 2-11 show the average surface concentrations of chlorophyll-a, total phosphorus, and SRP during the summer season (May through September) in the transitional section of each lake. The chlorophyll-a concentrations in the transitional sections of Lakes Hugo, Sardis, and Tenkiller during 2003 and 2005, where contemporaneous data are available, were similar in magnitude. These transitional section concentrations in 2003 and 2005 were 9.2, 7.0, and 8.0 µg/L and 11.0, 7.4, and 15.6 µg/L, respectively. Similar transitional section chlorophyll-a concentrations indicate that despite the smaller poultry populations in the Hugo and Sardis watersheds, a shorter residence time in Lake Hugo, and the lower lake to watershed ratio of Lake Sardis, the three lakes exhibit similar potential impact from their respective watersheds. This conclusion is further supported by similar total phosphorus and SRP concentrations in the transitional sections of the three lakes from 2003 and 2005. The average transitional section total phosphorus concentrations in Lakes Hugo, Sardis, and Tenkiller in 2003 and 2005 were 0.08, 0.02, and 0.17 mg/L and 0.08, 0.03, and 0.02 mg/L, respectively. The average transitional section SRP concentrations in 2003 and 2005 were 0.03, 0.01, and 0.09 mg/L and 0.03, 0.01, and <0.01, respectively.

Table 2-11. Summer surface average and chlorophyll-a, total phosphorus, and SRP concentrations in the transitional section of Lakes Hugo, Sardis, and Tenkiller from 2003 and 2005.

Parameter	Year	Lake	Number of Observations	Average	Minimum	Maximum	Units
Chlorophyll-a	2003	Hugo	5	9.2	4.9	13.0	mg/L
		Sardis	2	7.0	6.7	7.3	mg/L
		Tenkiller	13	8.0	2.9	33.2	mg/L
	2005	Hugo	3	11.0	8.0	13.0	mg/L
		Sardis	2	7.4	6.4	8.4	mg/L
		Tenkiller	25	16.2	8	32.3	mg/L
Total Phosphorus	2003	Hugo	6	0.077	0.068	0.091	mg/L
		Sardis	4	0.017	0.010	0.023	mg/L
		Tenkiller	5	0.171	0.025	0.310	mg/L
	2005	Hugo	6	0.081	0.072	0.093	mg/L
		Sardis	2	0.028	0.027	0.028	mg/L
		Tenkiller	15	0.023	0.003	0.033	mg/L
Soluble Reactive Phosphorus	2003	Hugo	6	0.031	0.016	0.043	mg/L
		Sardis	4	0.009	0.007	0.011	mg/L
		Tenkiller	6	0.090	0.010	0.190	mg/L
	2005	Hugo	6	0.033	0.024	0.043	mg/L
		Sardis	2	0.011	0.010	0.011	mg/L
		Tenkiller	15	0.002	0.001	0.005	mg/L

An analysis of the average surface concentrations of chlorophyll-a, total phosphorus, and SRP during the summer season (May through September) was also performed for the lacustrine section of the three lakes (Figure 2-20 and Table 2-12). Generally, the total phosphorus and SRP concentrations are lower in the lacustrine section of each lake as compared to upstream sections due to the settling of nutrients to the lake floor and phytoplankton utilization of the nutrients upstream of the lacustrine section. Chlorophyll-a concentrations are generally lower in the lacustrine section because nutrient concentrations are lower.

Table 2-12. Summer surface average total phosphorus, SRP, and chlorophyll-a concentrations in the lacustrine section of Lakes Hugo, Sardis, and Tenkiller from 2003 and 2005.

Parameter	Year	Lake	Number of Observations	Average	Minimum	Maximum	Units
Chlorophyll-a	2003	Hugo	4	5.5	2.6	10.6	µg/L
		Sardis	3	5.8	4.5	6.5	µg/L
		Tenkiller	26	4.8	1.2	9.9	µg/L
	2005	Hugo	2	9.0	6.0	12.0	µg/L
		Sardis	2	7.3	6.6	8.1	µg/L
		Tenkiller	47	11.1	4.0	36.8	µg/L
Total Phosphorus	2003	Hugo	4	0.060	0.040	0.081	mg/L
		Sardis	6	0.017	0.010	0.037	mg/L
		Tenkiller	13	0.146	0.011	0.420	mg/L
	2005	Hugo	4	0.068	0.051	0.090	mg/L
		Sardis	4	0.022	0.005	0.028	mg/L
		Tenkiller	24	0.013	0.008	0.027	mg/L
Soluble Reactive Phosphorus	2003	Hugo	4	0.027	0.016	0.038	mg/L
		Sardis	6	0.008	0.006	0.010	mg/L
		Tenkiller	17	0.073	0.005	0.170	mg/L
	2005	Hugo	4	0.031	0.019	0.046	mg/L
		Sardis	4	0.009	0.005	0.010	mg/L
		Tenkiller	22	0.003	0.001	0.013	mg/L

Identical comparisons and trends were apparent in the lacustrine sections as in the transitional sections of the three lakes. The 2003 and 2005 average summer surface chlorophyll-a and nutrient concentrations in the lacustrine sections of the three lakes were similar and lacustrine section nutrient concentrations in Lake Tenkiller decreased from 2003 to 2004. Lakes Hugo, Sardis, and Tenkiller average 2003 and 2005 summer surface chlorophyll-a lacustrine section concentrations were 5.5, 5.8, and 4.8 µg/L and 9.0, 7.3, and 11.1 µg/L, respectively. Lacustrine section total phosphorus concentrations were 0.06, 0.02, and 0.15 mg/L and 0.07, 0.02, and 0.01 mg/L, and SRP concentrations were 0.03, 0.01, and 0.07 mg/L and 0.03, 0.01, and <0.01 mg/L, respectively. These results further indicate similar water quality in the three lakes despite the lower poultry populations in the Hugo and Sardis watersheds and existing conditions in Lakes Hugo and Sardis that should improve water quality as compared to Lake Tenkiller (shorter residence time and lower watershed to lake ratio, respectively).

Figure 2-21 shows dissolved oxygen profiles in the lacustrine sections of Lakes Tenkiller, Hugo, and Sardis and the Plaintiff's comparison reservoir, Lake Broken Bow. The profiles were all taken during July and August. The four lakes have relatively high dissolved oxygen concentrations in the top 5 m and then trend toward zero dissolved oxygen near 10 m depth. Data are not available in Lake Sardis below 8 m and Lake Hugo has a relative shallow average depth, but the trend of the data appears similar for all four reservoirs. These dissolved oxygen profiles indicate that all of the reservoirs experience the common phenomena of dissolved oxygen depletion, even those whose watersheds have little poultry population. In fact, Sardis and Broken Bow, which have significantly lower watershed to water surface area ratios than the Tenkiller and Hugo, and thus potentially lower nutrient impacts, still show oxygen depletion in the bottom waters. In fact, most man-made run-of-river reservoirs will experience some level of dissolved oxygen depletion in the bottom waters, unless some other mechanism (such as wind mixing in shallow reservoirs) hinders dissolved oxygen depletion. In general, altering a natural system via dam construction inevitably results in water quality issues. Consequently, thermal stratification and resulting low dissolved oxygen levels in deeper waters is normal for run of the river reservoirs (Thornton et al. 1990)

Finally, chlorophyll-a TSI values were calculated for each section and the entire lake of Lakes Tenkiller, Hugo, and Sardis for the summer of 2005 (Figure 2-22). Trophic State Index provides a "rule-of-thumb" measure of the trophic status of the reservoir. The Oklahoma Water Resources Board (OWRB) uses chlorophyll-a TSI to assess what lakes in Oklahoma are eutrophic (or hypereutrophic) and potentially need to be managed to control algae. The TSI values calculated from a compilation of all available data are similar to the values found in Oklahoma's Beneficial Use Monitoring Program (BUMP) – Lakes Sampling, 2006-2007 Draft Report (OWRB 2007; eutrophic or borderline eutrophic). These results further support the existence of similar water quality issues in the three lakes, regardless of their poultry populations or conditions in Lakes Hugo and Sardis that should mitigate water quality impacts (shorter residence time and lower watershed to lake ratio, respectively).

Water quality issues in watersheds with low poultry populations relative to the Illinois River Watershed supports the conclusion that poultry litter is not the primary reason for water

quality issues that exist in Lake Tenkiller. There are other factors affecting water quality in Lakes Tenkiller, Hugo, and Sardis. These include:

1. urban and rural development which increases impervious cover, lawn and golf course fertilization, wastewater treatment plant (WWTP) discharges, and the number of septic systems in the watershed (Nelson et al. 2002; Soerens 2003; Sonoda 2007);
2. deforestation and related erosion (Perry et al. 1999; Zheng 2005; Grip 2008; Grip 2009);
3. row crop synthetic fertilizers and related erosion (Sharpley and Smith 1990; Sharpley et al. 2003; Wortmann 2005);
4. other livestock operations such as cattle and swine (USDA 2003; Shaffer 2005; Wortmann 2005; Beede 2007); and
5. inputs from humans during recreational use (see Jarman 2008 for discussion).

Finally, and most importantly, altering a natural system via dam construction inevitably results in water quality issues. These water quality issues arise due to restricting sediment flux out of a watershed and decreasing the potential and kinetic energy of the system, which increases residence time in the water body and thus promotes growth of phytoplankton.⁸

2.9 WASTEWATER TREATMENT PLANTS APPEAR TO BE THE MOST IMPORTANT SOURCE OF BIOAVAILABLE PHOSPHORUS TO THE SYSTEM

Many wastewater treatment plants in the Arkansas and Oklahoma portions of the Illinois River Watershed installed significant upgrades within the past decade, the majority of which were in place by 2004 (Jarman 2008). Improvements have been seen in water quality

⁸ Lakes Hugo and Sardis watersheds do not have significantly more urbanization, human population, or other animal populations compared to Lake Tenkiller. Consequently, the water quality issues observed in Lakes Hugo and Sardis even with the lower poultry populations can not be attributed to just urbanization, deforestation, or other animal populations.

immediately downstream of these facilities, and in some cases the water quality improvements have been noted far downstream in the wider Illinois River Watershed.

Wastewater treatment plants and their impact on Illinois River waters have been studied for numerous years. Haggard et al. (2003) and Ekka et al. (2003) indicate that base flow concentrations of phosphorus were elevated for streams receiving WWTP discharges. Haggard (2005) attributes decreased dissolved phosphorus concentrations in Spring Creek, and downstream in Osage Creek and the Illinois River, to upgrades to the Springdale municipal WWTP. Arkansas Department of Environmental Quality (ADEQ; 2008a) notes decreases in phosphorus concentrations in Siloam Springs, Sager Creek, and Little Sugar Creek over the past decade, in conjunction with treatment plant upgrades. Arkansas Water Resources Center (AWRC 2007) associated reduced total phosphorus base flow loads downstream of Siloam Springs to reduced wastewater treatment plant effluent loads, and found a strong correlation.

WWTP impacts continue to be seen in the water bodies in the Illinois River Watershed. Twenty-two percent of the impaired water bodies in the Oklahoma portion of the watershed include ‘municipal point sources’ as potential causes of the impairment (ODEQ 2008). 8.1 miles of Sager Creek remain impaired due to municipal point sources (ADEQ 2008b).

There are nine notable WWTPs that discharge to the streams of the Illinois River Watershed. Three are in Oklahoma and six are in Arkansas. Information about these plants is presented in Table 2-13.

Table 2-13. Wastewater treatment plants discharging to the Illinois River Watershed.

Plant	State	Receiving Water	Connection to the Illinois River	Average Total Phosphorus Load 2004 to 2007 (kg/yr)
Prairie Grove	AR	Unnamed tributary of Muddy Fork	Muddy Fork	2,000
Fayetteville – West	AR	Mud Creek (2004 – June 2007) Goose Creek (July 2007 – present)	Clear Creek Goose Creek	2,300
Springdale	AR	Spring Creek	Osage Creek	11,300
Rogers	AR	Osage Creek	Osage Creek	5,700
Siloam Springs	AR	Sager Creek	Flint Creek	13,000
Tahlequah	OK	Tahlequah Creek	Tahlequah Creek	1,200
Lincoln	AR	Unnamed tributary of Bush Creek	Baron Fork	270
Westville	OK	Shell Branch of Baron Fork	Baron Fork	330
Stillwell	OK	Caney Creek	Caney Creek	900

In total, over the period from 2004 to 2007 these plants discharged an average of almost 37,000 kg of phosphorus per year to the streams of the Illinois River Watershed, not counting any spikes in discharge that may have occurred due to plant upsets or short-circuiting during storm events (Jarman 2008). Much of the phosphorus entering the streams from these plants is dissolved and most of the dissolved phosphorus is reactive (i.e., SRP), the form that stimulates plant growth. This fact is evident in Figure 2-23, which shows the fraction dissolved and fraction of dissolved that is SRP for phosphorus measurements conducted by the Plaintiffs on WWTP effluent.

The influence of the WWTPs is evident in the spatial pattern of phosphorus concentrations in the rivers and streams of the Illinois River Watershed, as shown in Figure 2-24a. The highest total phosphorus concentrations (typically red or orange symbols) are found almost always just downstream of WWTPs (yellow diamonds in the figure). Moving further downstream there is typically a downward trend in concentrations indicated by the transition to green, light blue and finally dark blue symbols. High concentrations occur at a few stations remote from WWTPs, but the only organized spatial patterns are tied to the WWTPs.⁹ A similar pattern exists for SRP, which is shown in Figure 2-24b.

⁹ The location of the wastewater treatment facility in Watts, OK is also indicated on these figures. This is a retention and land application facility and is not permitted to discharge, however at least one release is documented (Jarman 2008). Sampling data from the Illinois River immediately downstream of the Watts facility is not available,

A more quantitative examination of the spatial patterns is presented in Figure 2-25, which shows the upstream to downstream trend in SRP concentrations in the Illinois River for three time periods (1998 to 2000; 2001 to 2003; 2004 to 2008). Red arrows indicate the locations where major tributaries enter the Illinois River. Moving from upstream to downstream, there is a gradual increase in SRP concentration from levels less than 0.01 mg/L to about 0.03 mg/L just above Muddy Fork (data only in the 2004-2008 period). The two sampling locations between Muddy Fork and Osage Creek exhibit similar concentrations in the range of 0.03 to 0.05 mg/L. The first station downstream of Osage Creek has concentrations in the neighborhood of 0.15 mg/L, a substantial increase from the nearest upstream station. This increase suggests that Osage Creek is an important source of SRP to the Illinois River. The reach between Osage Creek and Lake Frances shows increases in the two earlier time periods (though not statistically significant) and a statistically significant¹⁰ decrease in the latest period. Concentrations generally decline between Lake Frances and Lake Tenkiller reaching about 0.07 to 0.09 mg/L just above Lake Tenkiller. The locations where these samples were collected are identified on Figure 2-26.

Given the apparent importance of Osage Creek, the spatial pattern in this creek and its tributaries was examined. Focusing on the 2004-2008 period (Figure 2-27), which has the best spatial coverage, and August 2006 (Figure 2-28) to provide a synoptic view, it is apparent that the influence of Osage Creek on SRP in the Illinois River is due to WWTPs. Beginning on Spring Creek, SRP concentrations are less than 0.1 mg/L upstream of the Springdale WWTP and about 0.45 mg/L just downstream of the plant. On average, levels decline to about 0.2 mg/L just upstream of the confluence with Osage Creek, although they are at 0.35 mg/L in August 2006. In Osage Creek, the concentration is about 0.01 mg/L upstream of the Rogers WWTP and 0.25 mg/L downstream of the plant. There is a drop to about 0.15 mg/L just upstream of the confluence with Spring Creek and an increase to close to 0.2 mg/L downstream of the confluence. Just above the confluence with Illinois River the concentration is about 0.12 mg/L (0.2 mg/L in August 2006). Similar patterns are shown in data measured before 2004

¹⁰ Statistical significance inferred when differences fall outside the 2 standard error range indicated by the error bars around the mean values.

(Figure 2-29), indicating that historically, WWTP discharges had an influence on the phosphorus concentrations in the rivers and streams. See Figure 2-26 for sampling locations.

Wastewater treatment plants impact phosphorus concentrations in the Illinois River every day, whereas most other sources (except perhaps septic tanks) contribute only during runoff events that occur periodically and somewhat infrequently during the summer season when phosphorus impacts water quality. In fact, the amount of phosphorus in the Illinois River under base flow conditions corresponds to the amount that entered upstream from WWTPs, indicating that the WWTPs are the dominant source of phosphorus during base flow. This correspondence is shown in Figure 2-30, which displays the distribution of base flow phosphorus loadings measured by the United States Geological Survey (USGS) in the river at monitoring stations at Watts and Tahlequah and shows as vertical lines the average daily loading from the WWTPs. The average load from the WWTPs matches the central tendency base flow load in the river. The variability in the river around the central tendency likely reflects the day-to-day variability in WWTP load.

The 2004-2006 average daily wastewater treatment plant total phosphorus loads were also compared to 2004-2006 Illinois River and tributary average daily total phosphorus loads under base flow (WWTP data for 2007 were incomplete, therefore 2007 is not shown). Available daily flow and total phosphorus data from USGS gauging stations at Watts, Tahlequah, Baron Fork, and Caney Creek were used to estimate average daily total phosphorus loads with LOADEST, a program that estimates average loads through a rating curve method (Runkel et al. 2004).¹¹ As shown in Figure 2-31, the wastewater treatment plant loads (per Jarman 2008) are reasonable matches to the base flow loads in 2005 and 2006. The treatment plant loads appear lower than the in-river base flow loads in 2004 when frequent and significant high flow events potentially biased the estimation of base flow (i.e., some high flows identified as base flows may have included surface runoff) and the elevated base flows may have introduced a greater load from septic systems (see Figure C-1 to note the high base flows in

¹¹ LOADEST estimated daily loads with available paired daily average flow and total phosphorus data. Daily average flow data were used because instantaneous flow data were not available at all locations. Daily average total phosphorus loads are averages of daily total phosphorus loads estimated by LOADEST. LOADEST load estimates were generated using the model's Method 8 and separate rating curves were produced for each year.

2004). Note, the locations labeled as Baron Fork and Caney Creek in Figure 2-31 refer to the points in the Illinois River where the Baron Fork and Caney Creek tributaries meet the Illinois River.

In contrast to base flow phosphorus, runoff-associated phosphorus is not present in the river on a day-to-day basis. In addition, much of the runoff phosphorus load is associated with particulate matter, which would have little direct impact on water quality (it can exert an influence via recycle from sediments). This fact is illustrated in Figure 2-32, which shows the fraction of total phosphorus that is particulate in relation to river flow (particulate phosphorus is calculated by subtracting dissolved phosphorus from total phosphorus). A consistent increase with increasing flow is evident.

The particulate phosphorus associated with runoff events will only settle out of the water column when the river velocity is less than about 15 miles/day (Ziegler et al. 2000). Due to the high velocities characteristic of the Illinois River within Oklahoma¹² (Figure 2-33), little of the particulate phosphorus settles in the river. Much of the runoff particulate phosphorus likely settles out in Lake Tenkiller. This sediment phosphorus might later contribute to phosphorus levels in the lake if it fluxes out of the sediment, but in general it has limited bioavailability (see section 2-10).

During the summer season (May to September), the river experiences runoff conditions only about 20% of the time.¹³ Due to the short duration of runoff events, their relative infrequency, and the nature of the phosphorus, run-off associated phosphorus has little impact on water quality, except possibly within Lake Tenkiller.

¹² River velocities determined using Manning's Equation with a Manning's roughness coefficient of 0.04; slope determined from USGS gage heights (when available) or Google map topographic elevations, and river distances determined from GIS using Environmental System Research Institute data; depths of water surface determined from USGS depth data coincident with average summer-time flow rates at each USGS gage location. Riverine portion of lake velocities determined by dividing summer-time average flow rate just downstream of Baron Fork by the approximated cross section of the riverine portion of the lake between Baron Fork and LK04; distance from Baron Fork to LK04 determined from GIS.

¹³ The contributions of base flow and runoff flow to the river hydrograph was determined using a base flow separation methodology described in Appendix C.

2.10 LAKE SEDIMENT PHOSPHORUS IS A MINOR SOURCE OF BIOAVAILABLE PHOSPHORUS

Phosphorus enters Lake Tenkiller with point source (i.e., WWTP) dominated base flow, non-point source dominated high flow, and groundwater discharge. The eventual fate of phosphorus once it enters the lake varies depending on the specific forms of phosphorus present. Any dissolved phosphorus will either remain in the water column of the lake or be discharged downstream via the dam. SRP can be taken up by algae and eventually converted to particulate phosphorus. Particulate forms of phosphorus may remain in the water column, be discharged downstream via the dam, or settle to the bottom and become incorporated in the sediment.

Phosphorus movement into, through, and out of the reservoir is illustrated as a conceptual diagram (Figure 2-34). Of note is the summertime stratification of the reservoir into 2 layers. Reservoirs such as Lake Tenkiller tend to be completely mixed, with similar temperature and chemical constituents throughout, during the winter (Lewis 1983). The warming of the water surface during early spring initiates separation of the lake water into distinct layers. As the temperature of the surface water exceeds 39°F, its density declines.¹⁴ This more buoyant water remains near the top of the water column. Warm, bouyant water toward the lake surface becomes the summer epilimnion, or top layer of the water column. Dissolved oxygen levels increase due to exposure to the atmosphere (USEPA 2000). Light is available for photosynthesis, and the potential for further increases in dissolved oxygen. The presence of algae will be indicated by increased levels of chlorophyll-a.

The colder, denser layer that forms at the bottom of the lake is called the hypolimnion, where low temperature and lack of light penetration, inhibit algae growth. Due to differences in density, these two layers do not mix, and there is little transport of dissolved chemical constituents (including oxygen) between the epilimnion and the hypolimnion.

With colder weather in the autumn, the epilimnion water cools, and increases in density. When the density of the epilimnion water exceeds the density of the hypolimnion, a fall turnover

¹⁴ Water reaches maximum density at 39°F. Above or below this temperature, water will be more buoyant.

occurs; epilimnion waters tend to sink and hypolimnion waters tend to rise. The temperature and density of the water is closer to uniform, so individual layers do not remain after the turnover. The lake remains completely mixed during the winter, and the cycle repeats in the spring.

Since 2004, about 205,000 kg/yr total phosphorus entered Lake Tenkiller from the Illinois River, Baron Fork, and Caney Creek (Bierman 2009). About 30,000 kg/year exited via Tenkiller Dam.¹⁵ The remaining 175,000 kg/year of phosphorus was incorporated into the bottom sediments, with minimal if any impact on algae levels in the lake, as described below.

During the summer months, dissolved oxygen is depleted in the hypolimnion. When dissolved oxygen is very low or zero in the hypolimnion, some of the phosphorus in the sediments can return to the water column as dissolved phosphorus, largely in the form of SRP. This flux increases the summer SRP concentration in the hypolimnion, and contributes phosphorus to the surface waters when the lake overturns in late fall. In Lake Tenkiller, the hypolimnetic SRP mass increases by approximately 3,000 kg¹⁶ during the summer, but this mass is not large enough to have a material impact on the epilimnion SRP concentrations when the lake turns over (top row of graphs in Figure 2-35); SRP concentrations never get higher than 0.01 mg/L in the epilimnion.

It should be noted that Figure 2-35 shows that *total* phosphorus increases in the hypolimnion during the summer. In fact, this increase is quite significant compared to the increase in SRP and other forms of dissolved phosphorus (not shown), indicating that the increase in phosphorus is comprised mostly of particulate-bound phosphorus. This is likely the result of river water plunging in the vicinity of LK-04 and “pulling” chlorophyll-a and suspended sediment from the surface waters into the bottom waters (see Section 4.2 for further discussion of the plunging river water in the vicinity of LK-04). It is not likely caused by resuspension because one would expect a more random or event-based (i.e., scour events) signature.

¹⁵ Estimated using a long-term mean of hypolimnion phosphorus concentrations and the 1994 – 2007 United States Army Corps of Engineers record of water release at the dam.

¹⁶ Mass of SRP in the hypolimnion was found using the Plaintiffs’ data collected in the deep waters of Lake Tenkiller, combined with an estimate of the hypolimnion volume from Dr. Wells’ lake model bathymetry.

Although non-point-source phosphorus loads contribute to sediment phosphorus in Lake Tenkiller, much of this phosphorus is locked in the sediment and does not contribute to algae growth.



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MEMORANDUM

To:	File	Date:	May 7, 2009
From:	John Connolly, Anchor QEA	Project:	OICiln:114
Cc:			
Re:	Errata to Expert Report: Illinois River Watershed Water Quality and Source Assessment		

Documented herein are corrections made to Figures 2-30, 2-31, 2-32, 7-1, and 7-2 in my 2008 expert report for the Defendants in the case of the State of Oklahoma against the Illinois River Watershed Joint Defense Group. This information is submitted in response to questions raised in my deposition and to correct data sources in Figure 2-30.

Figures 2-30 and 2-31 of my report intended to compare phosphorus loads at specific Illinois River United States Geological Survey (USGS) stations to combined phosphorus loads from WWTP upstream of these stations. The phosphorus load from the Tahlequah wastewater treatment plant (WWTP) was mistakenly included in the combined WWTP load upstream of USGS station 07196500 (the Tahlequah WWTP input is downstream of USGS station 07196500). In the amended Figures 2-30 and 2-31, the Tahlequah WWTP phosphorus load has been removed from the combined WWTP phosphorus load upstream of USGS station 07196500.

Figure 2-30 has been corrected to remove the two highest daily base flow phosphorus loads from the phosphorus loads measured at Tahlequah USGS station 07196500. Upon review of the USGS flow data and the United States Army Corps of Engineers precipitation data, these two loads were found to have occurred in the midst of, or immediately following, extended high precipitation periods (Figure 1) and were therefore consistent with runoff rather than base flow conditions. In Figures 2-31, 2-32, 7-1, and 7-2 these same two highest base flow loads were switched from the base flow to the high flow loads to place them in the correct dataset.

The caption for Figure 2-30 has been corrected to specify the date ranges for all four source data sets (OWRB, ADEQ, USGS, and the Plaintiff database). The previous caption only specified the date range for the Plaintiff database.

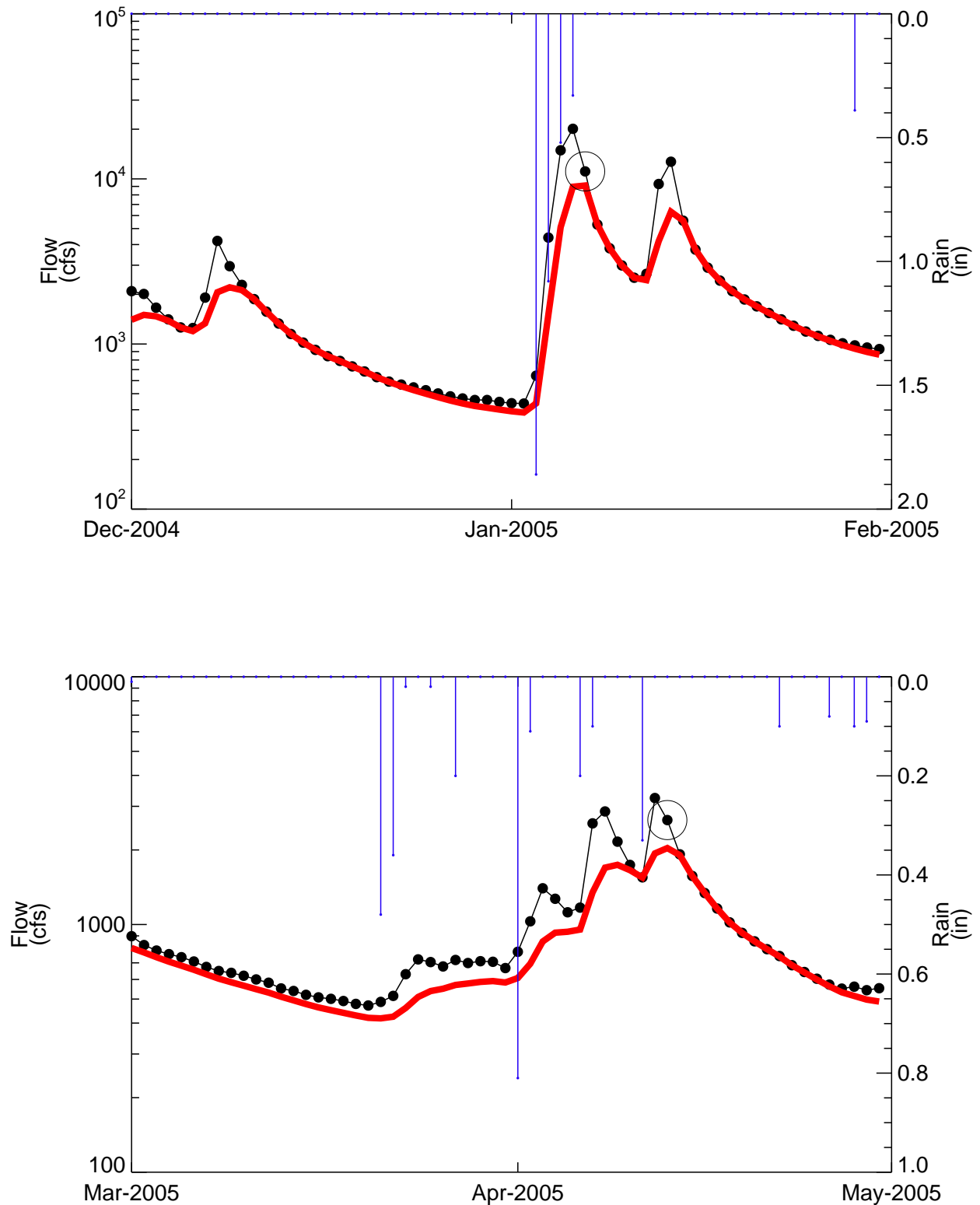
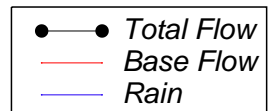


Figure 1. Total flow, base flow and precipitation at Tahlequah (Dec 2004 - Feb 2005; Mar 2005 - May 2005).

*The circled points represent dates 1/7/2005 and 4/13/2005.
Data source: USGS*



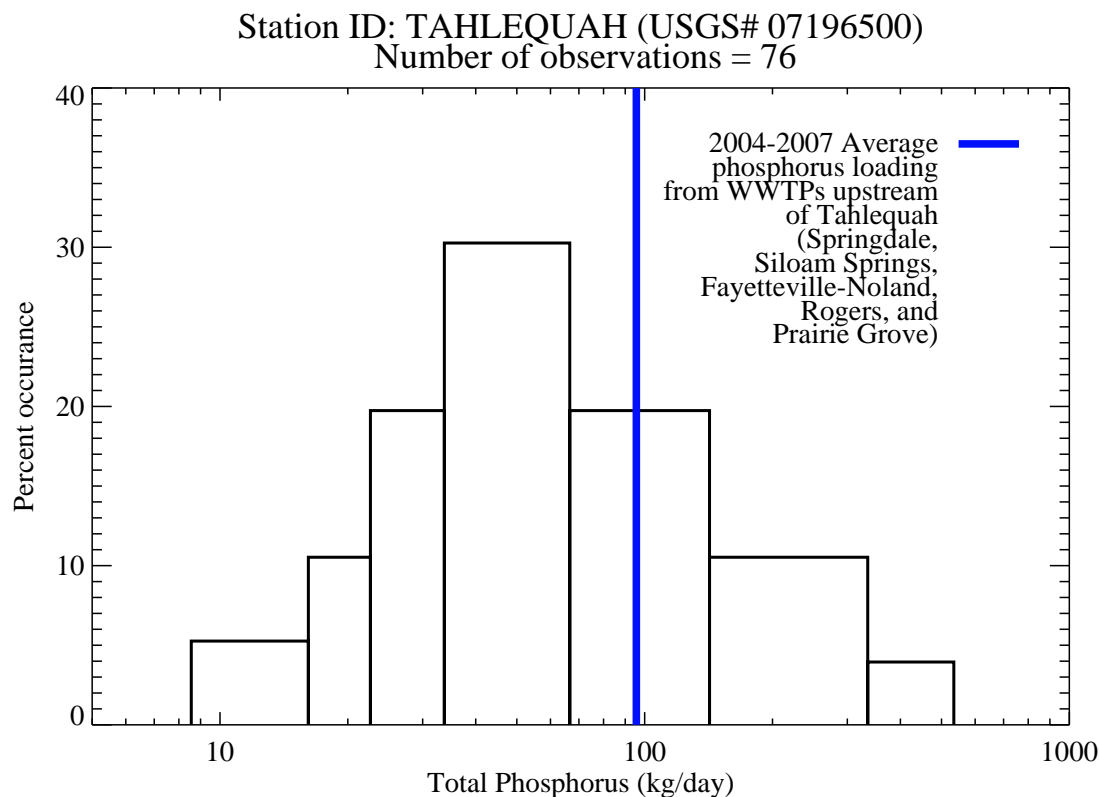
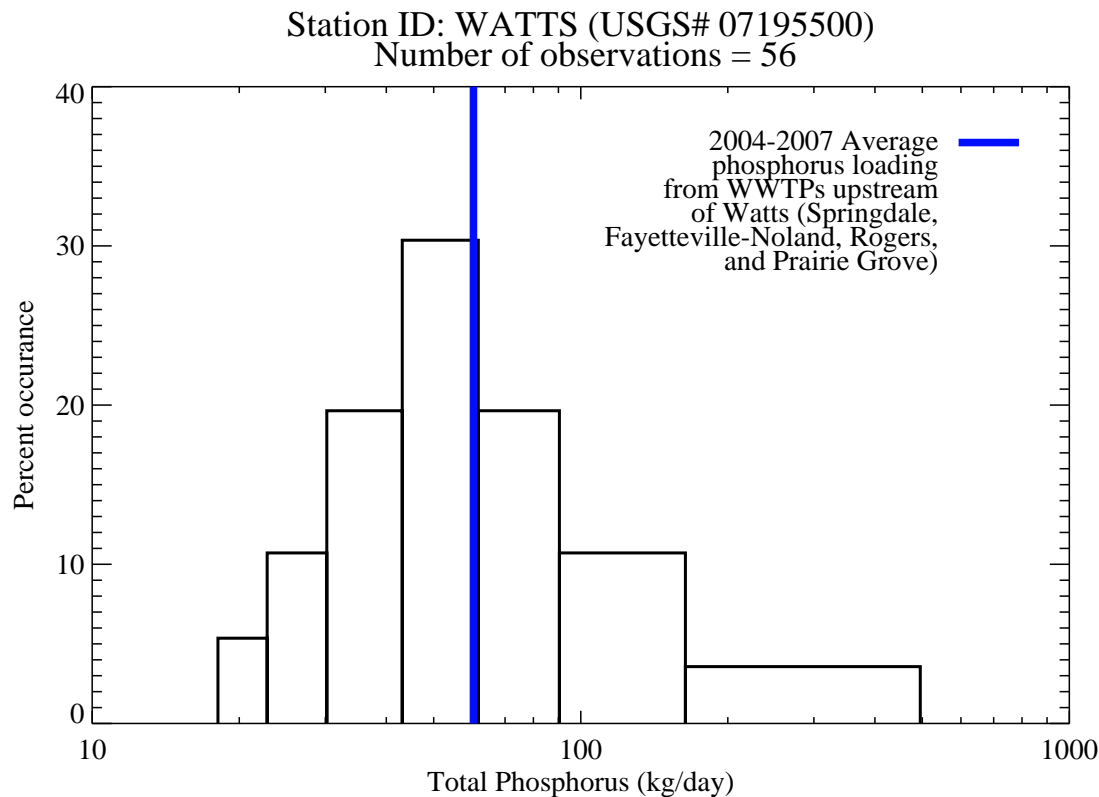


Figure 2-30. Frequency of calculated loads for total phosphorus under base flow condition, 2004-2007.

WWTP loads from Jarman 2008.

River loads calculated for days that had measured flow (USGS) and phosphorus concentration data (OWRB, ADEQ and USGS data collected 2004-2007, Plaintiff's data collected 2005-2007). The two highest load values at Tahlequah were removed due to occurrence during flows too high to be considered base flow.

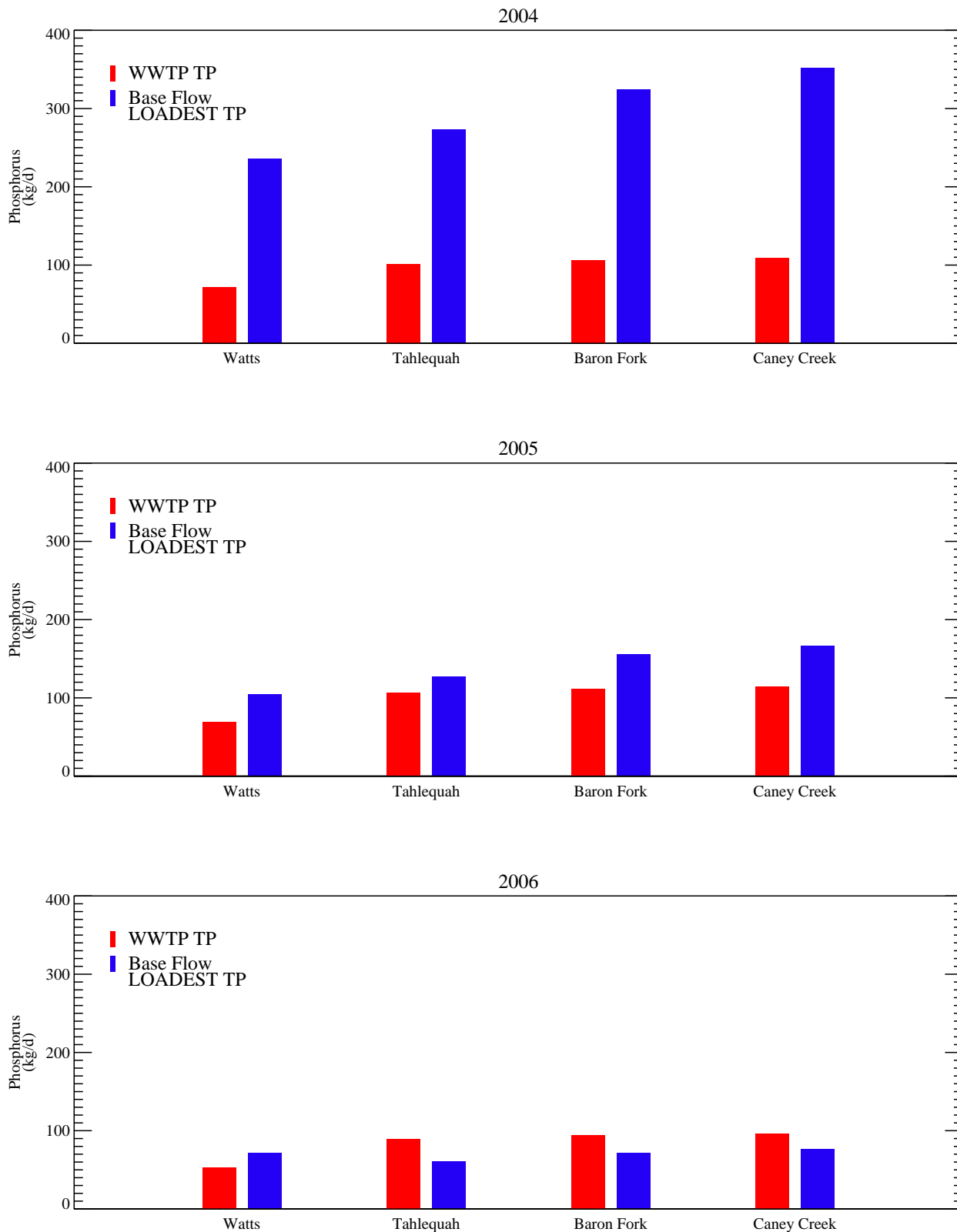


Figure 2-31. Average daily phosphorus loads along Illinois River.

Locations: USGS stations at Watts and Tahlequah; points of confluence of Baron Fork and Caney Creek with Illinois River. Values plotted at each location are cumulative loads till that point on the river. LOADEST calibrated for 2004-2008. Data sources in text.

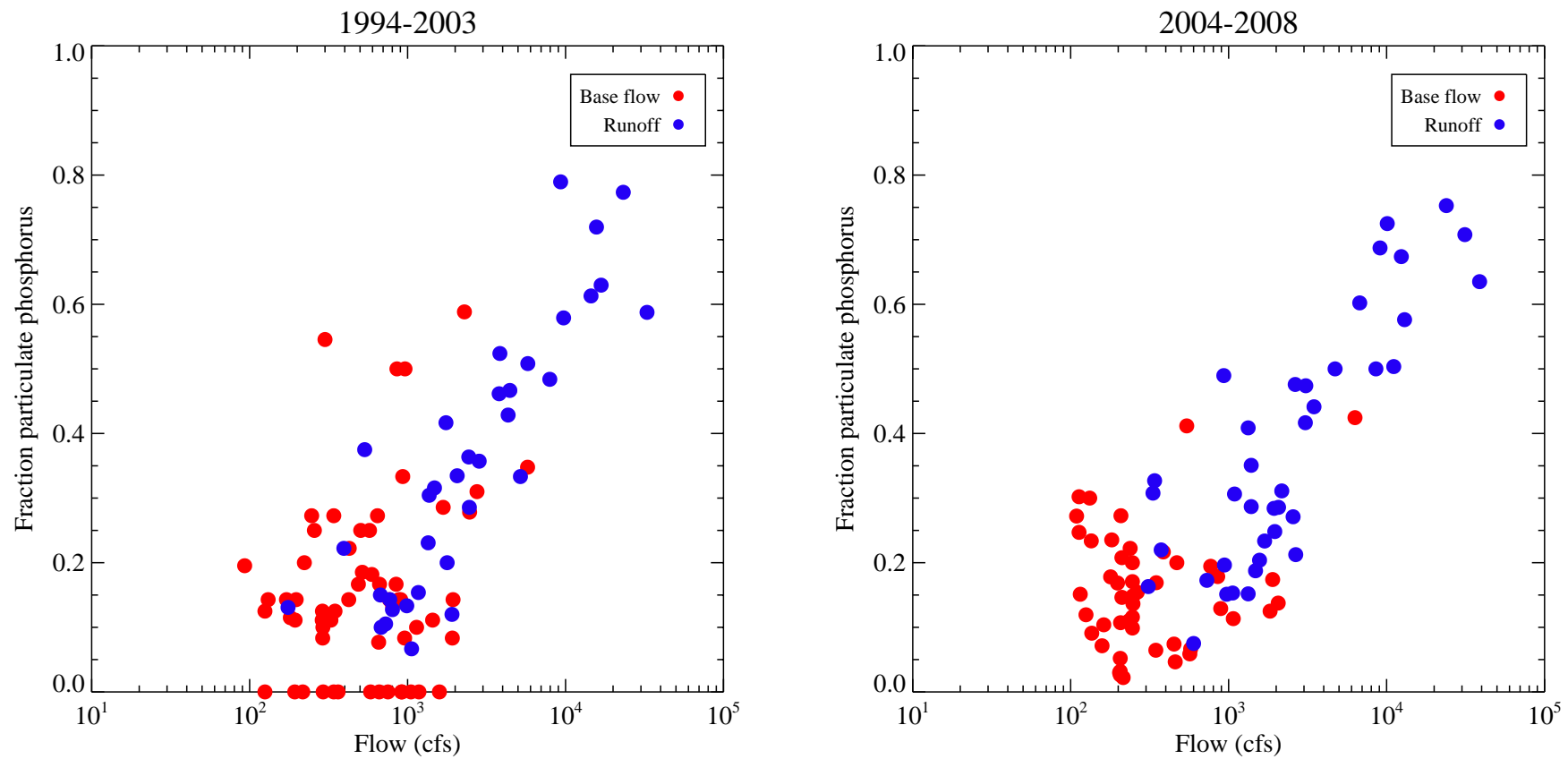


Figure 2-32. Fraction particulate phosphorus as a function of flow at Tahlequah.
Data: Plaintiff's Database 2004-2008, USGS 1994-2008.

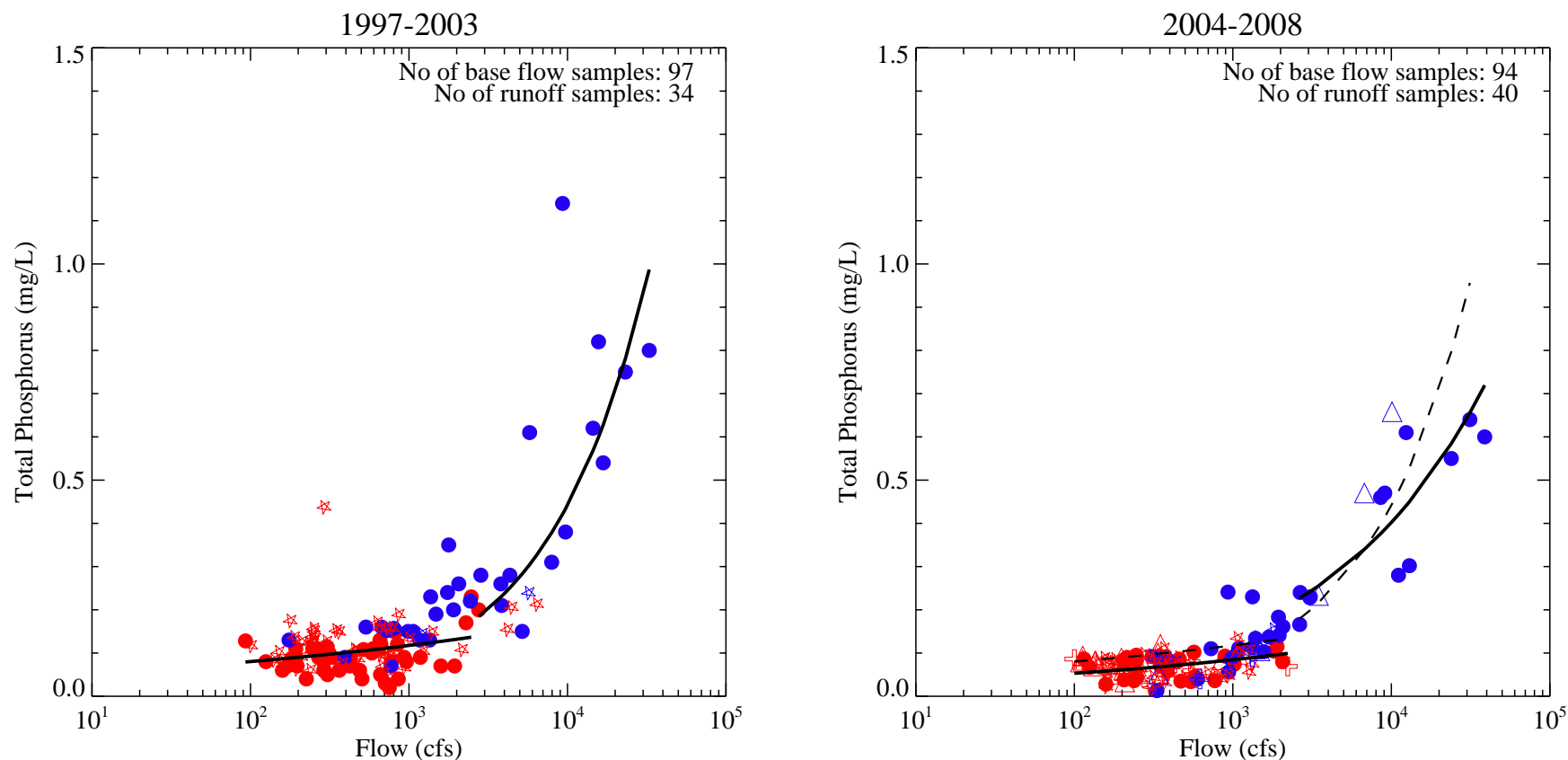


Figure 7-1. Total phosphorus concentrations as a function of flow at Tahlequah.

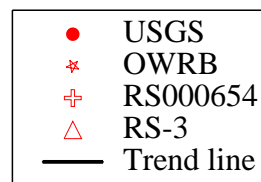
Red points are base flow conditions, blue points are runoff conditions.

Regression lines are least square estimates for flows ≤ 2500 cfs and flows > 2500 cfs.

The dashed line in the second panel represents 1997-2003 trend line for comparison.

Base flow conditions are days when base flow is 70% or greater of total flow.

Data: USGS 1997 - 2008, Plaintiff's Database 2004-2008, OWRB 1998 - 2008.



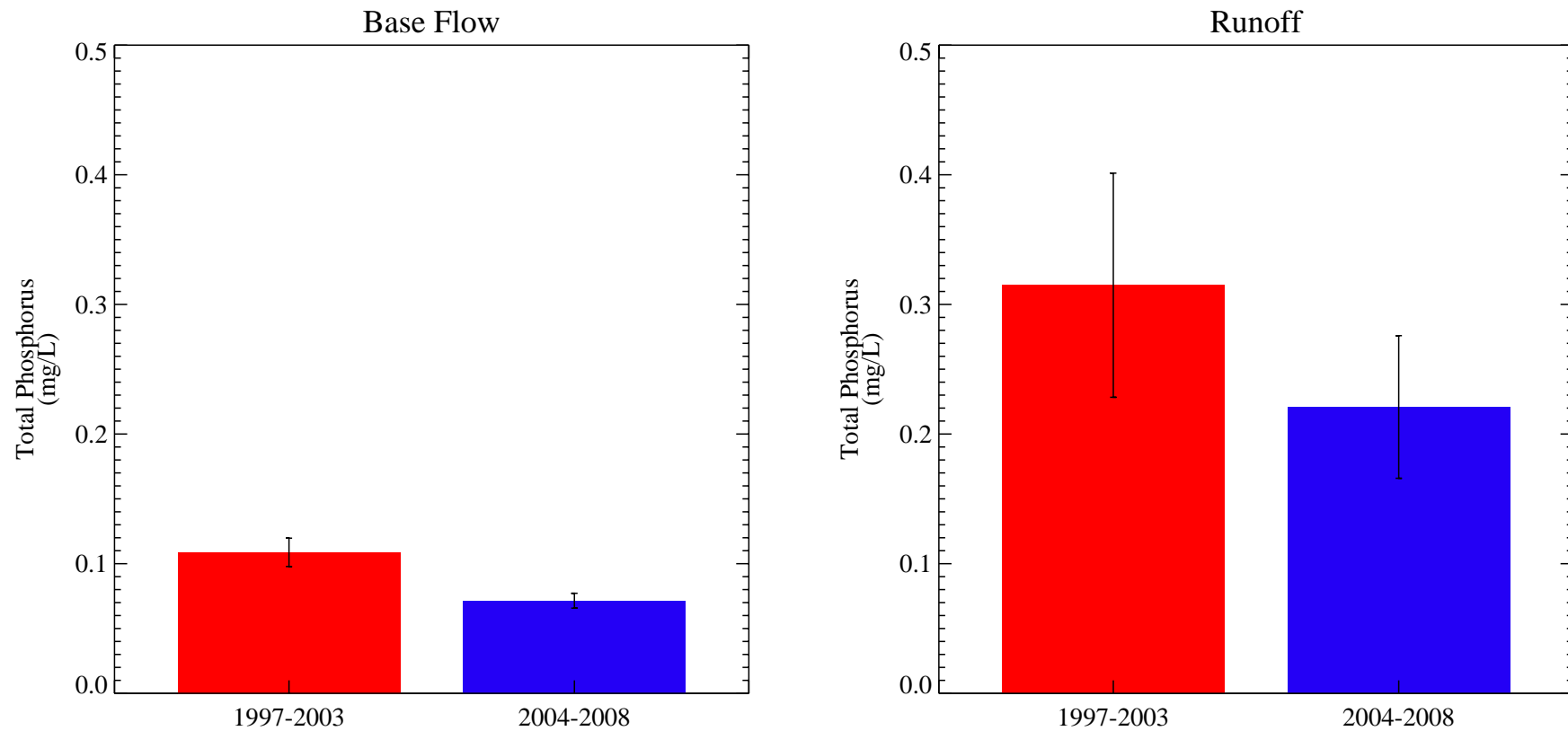


Figure 7-2. Average total phosphorus during base flow and run off events at Tahlequah.

Error bars at +/- 2 standard errors.

Data: USGS 1997-2008, Plaintiff's Database 2004-2008, OWRB 1998-2008.